

THE INFLUENCE OF FIRE AND OTHER DISTURBANCE ON ERICACEOUS  
SHRUBS IN XERIC PINE-OAK FORESTS OF THE  
APPALACHIAN MOUNTAINS

A Thesis

by

ASHLEY R. PIPKIN

Submitted to the Office of Graduate Studies of  
Texas A&M University  
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2011

Major Subject: Geography

The Influence of Fire and Other Disturbance on Ericaceous Shrubs in Xeric Pine-Oak  
Forests of the Appalachian Mountains  
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Approved by:

Chair of Committee,	Charles Lafon
Committee Members,	David Cairns
	Fred Smeins
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## ABSTRACT

The Influence of Fire and Other Disturbance on Ericaceous Shrubs in Xeric Pine-Oak  
Forests of the Appalachian Mountains. (May 2011)

Ashley R. Pipkin, B.A, University of South Carolina

Chair of Advisory Committee: Dr. Charles W. Lafon

Fire suppression in the southern and central Appalachian Mountains has resulted in an alteration to vegetation structure and composition. For this research the dominant species, abundance, density and age structure of the ericaceous shrub layer is characterized on four sites across the southern and central Appalachian Mountains. Fire histories for each of the sites varied, and were determined in previous research using dendroecological techniques. Over 800 ericaceous shrubs were collected. Species included *Pieris floribunda* (Pursh) Bentham and Hooker f., *Rhododendron maximum* L. and *Kalmia latifolia* L.. Basal area of ericaceous shrubs was significantly different between sites. Age structures show that when fire suppression started *Ericaceae* began to establish. A few *Ericaceae* cross-sections displayed scars, which are likely associated with fire events, suggesting they probably survived mild fire events. Ericaceous shrub age structures were compared with Southern Pine Beetle (SPB) outbreaks and the Palmer Drought Severity Index. There were no significant correlations, but field observations suggest that SPB may be providing conditions suitable for *Ericaceae* establishment. Topographic patterns reveal that *Kalmia latifolia* is most abundant at

mid-slope positions and decreases at higher and lower slope positions. There were significant differences in the density between slope positions averaged across all sites. Transects with the most recent and frequent fires did not have any of the three ericaceous shrubs collected at the slope bottom or ridge-top. At the most fire-suppressed site Ericaceae are present at every slope position. Age structures reveal that the oldest Ericaceae are found at the mid-slope positions while the age of thickets appears to decrease away from the mid-slope position. This pattern suggests that Ericaceae are moving into slope positions where they were previously less abundant. Sites with the most recent frequent fire regime seem to have prevented Ericaceae from heavily inhabiting high and low topographic positions while also reducing the overall basal area and density of Ericaceae.

## DEDICATION

I dedicate this document to my Grandparents; James, Corine, Giovanni and Concietta and to my Parents. Mom and Dad thank you so much for all of your support. I could not be happier that I was yours. Your silliness, love and sensitivity have shaped my understanding of the world and made my appreciation of it greater.

To James, thank you for opening my eyes and making Jamestown an interesting place. I do not think I will ever meet another person that understood the landscape better and made it more fascinating. To Corine, thanks for showing me the importance of kindness and giving back, it has gotten me further than most things in life will. To Giovanni, thank you for showing me what true determination is and inspiring me to set my goals high. To Concietta, thank you for showing me the value of hard work and reminding me to never stop pushing myself. I love you all very much.

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## NOMENCLATURE

JMP	Statistical Software
PDSI	Palmer Drought Severity Index
SPB	Southern Pine Beetle
spp.	Species
SPSS	Statistical Software



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## CHAPTER I

### INTRODUCTION

#### Background

Shrub expansion is occurring in multiple ecosystems across the world. Most shrub expansion research has taken place in arid and semi-arid non-forested regions (Van Auken 2000) but shrub expansion is also occurring in forested landscapes in temperate and boreal forests (Mallik 2003). Cover of ericaceous shrubs, or shrubs belonging to the family Ericaceae, has increased in forests. Increases in ericaceous shrubs have been attributed to the absence of high severity fire, the lack of overstory seeds able to germinate in dense ericaceous humus, and the ability of Ericaceae to degrade soil quality and quickly regenerate by vegetative methods (Mallik 2003). Without the frequent fire disturbances some Appalachian Mountain communities require, overall diversity will decline as shrub thickets continue to expand (Baker and Van Lear 1998).

This research uses dendroecological techniques to investigate the history of ericaceous shrub establishment in xeric pine and pine-oak forests of the Appalachian Mountains. Currently these shrubs, particularly mountain laurel (*Kalmia latifolia* L.), form dense thickets in the understory. It is hypothesized that shrub cover has expanded over the twentieth century as a consequence of fire exclusion (Dobbs and Parker 2004).

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This thesis follows the style of *Annals of the Association of American Geographers*.

Historically these stands burned frequently, at intervals ranging from 2-15 years (Harmon 1982; Shumway et al. 2001; DeWeese 2007; Hoss et al. 2008; Aldrich et al. 2010), likely maintaining more open conditions in the canopy and the shrub layer (Harrod et al. 2000).

A decline in fire frequency began under fire protection mandates starting in the early part of the 20<sup>th</sup> century (Brose et al. 2001 ). Fire exclusion has resulted in an increase in tree density, particularly shade-tolerant, fire sensitive species (Harrod et al. 1998; Lafon et al. 2005; Lafon et al. 2007) and promoted ericaceous shrub encroachment (Brose and Waldrop 2010). The expansion of ericaceous shrubs in the Appalachian Mountains has inhibited tree establishment in these disturbance-limited environments by reducing light, occupying space, producing a thick duff layer, consuming nutrients and reducing soil quality (Clinton and Boring 1994; Beckage et al 2000; Chastain and Townsend 2008; Mallik 2003).

Recent studies that report an increase in the ericaceous shrub layer demonstrate this through change detection using aerial and/or satellite mapping (Dobbs and Parker 2004), or short term vegetation surveys of the landscape (Elliott 1999). Both of these methods support the assertion that the shrub layer has expanded in the Appalachian Mountains. In addition to previous research a dendroecological study will help clarify temporal patterns of expansion and compare the history of fire and other disturbance events to shrub establishment.

Brose and Waldrop (2010) collected dendroecological data on ericaceous shrubs in the southern Appalachian Mountains, but did not collect individuals under 2.54 cm.

By ignoring this size class young Ericaceae are excluded which account for a large portion of the shrub layer and current regeneration is disregarded. Brose and Waldrop (2010) construct an age structure that shows the current shrub layer as having arisen after the last major fire and regenerating successfully without disturbance.

In this field based research over 800 shrubs were dated on four sites, with known fire histories, in the southern and central Appalachian Mountains. No previous studies have evaluated how fire history has impacted the current density and basal area of ericaceous shrubs. The data for this research were collected in xeric pine stands where dendroecological studies of fire history had already been completed (DeWeese 2007; Aldrich 2011; Laforest unpublished). These stands were composed primarily of fire intolerant yellow pine and oak species before fire suppression. Post fire suppression increases in sprouting shrubs and fire intolerant tree species have changed the vegetation dynamics on these sites.

### Objectives

The purpose of this study is to elucidate the dynamics of ericaceous shrub establishment under the changing fire regime. My specific objectives are to:

1. Describe the dominant species, abundance, density and age structure of Appalachian Mountain ericaceous shrub communities to reveal the current state of the ericaceous shrub layer and provide a context for investigating shrub establishment dynamics.



2. Determine temporal patterns of shrub establishment with respect to fire history.

The field data were collected on sites with known fire history (from previous dendroecological work).

3. Compare patterns of shrub expansion among different slope positions within each study site to explore the possibility that shrubs have expanded from particular parts of the landscape to occupy more topographic positions during this period of fire exclusion.
4. Examine how non-fire disturbances (insect outbreaks) and climatic variability may have contributed to shrub establishment.

By addressing the research questions above an understanding of the historical presence of fire and the current fire suppression regime can be reached. This study will have important implications for land managers tasked with controlling the increase and encroachment of ericaceous shrubs.

In addition to this introductory chapter this thesis includes five additional chapters. Chapter II is a literature review on shrub expansion and the disturbance regimes of the Appalachian Mountains. This literature review also describes both the shrubs collected in this study and their ecological role. Chapter III describes the methods used to address the research questions, how data were analyzed and introduces the study area. Chapter IV presents the results. Chapter V, the discussion, offers some interpretation of the results. Finally, Chapter VI presents conclusions that are suggested by the results.

## CHAPTER II

### LITERATURE REVIEW

#### Shrub Characteristics

Three ericaceous shrubs were collected in this research: *Pieris floribunda* (Pursh) Bentham and Hooker f., *Kalmia latifolia* L. and *Rhododendron maximum* L. (Figure 1). *Rhododendron maximum* occurs in low to mid elevations of the Appalachian Mountains and in mesic forests and heath balds (Whittaker 1962). This species is found along lower northeast facing slopes concentrated close to streams where it forms dense thickets (Baker and Van Lear 1998). *Kalmia latifolia* occurs at similar elevations but on more xeric sites, on ridges farther away from streams (Day and Monk 1974). All three species have sclerophyllous leathery leaves. Because of their high biomass both *Rhododendron maximum* and *Kalmia latifolia* have important influences on ecosystem function and processes (Monk et al. 1985). They are the most prevalent ericaceous shrubs found in low to mid elevation Appalachian Mountain forests (Whittaker 1962). *Pieris floribunda* has the smallest stature at around two meters with evergreen leaves (Luteyn 1996). *Pieris floribunda* and *Kalmia latifolia* share a similar habitat and look very similar except when in bloom. *Rhododendron maximum* has the largest stature followed by *Kalmia latifolia* and *Pieris floribunda*, respectively.



Figure 1: Ericaceous shrubs collected from study area. From left to right, a) *Pieris floribunda*, b) *Kalmia latifolia* and c) *Rhododendron maximum*. Pictures taken by Will Cook, Duke University.

Heath balds found in the Appalachian Mountains are composed of many species of Ericaceae with *Kalmia latifolia* and *Rhododendron spp.* accounting for the highest biomass. These communities are speculated to be the result of disturbance which destroyed the forest community allowing Ericaceous shrubs to take over (Whittaker 1956; Brose and Waldrop 2010). If Ericaceae are present in high volume when the canopy is destroyed they are in a position to increase their coverage and inhibit tree regeneration. Removal of forest dominants has caused an alteration in cover, which may eventually lead to areas dominated by shrub heathlands (Mallik 2003). These heath balds are stable communities which may prove difficult to convert back to forest species assemblages once established (Whittaker 1956).

*Rhododendron spp.* and *Kalmia latifolia* are often multi-stemmed (Figure 2) and grow in dense groupings often referred to as thickets (Hille Ris Lambers and Clark 2003). Clonal shrubs are capable of naturally producing potentially independent offspring by means of vegetative growth (Van Groenendael et al. 1996). Ericaceae can

reproduce from seed but more often thickets are formed from vegetative regeneration from a parent plant (Monk et al. 1985; McGee and Smith 1967; Lei et al. 2002). Mallik (1993) did a study on the vegetative growth forms of a congener *Kalmia angustifolia*. Because there have been no similar studies on *Kalmia latifolia* reference to Mallik (1993) is made when discussing vegetative growth. The growth forms described by Mallik coincide with my field observations and other research about *Kalmia latifolia* (McGee and Smith 1967; Lei et al. 2002). *Kalmia* can expand into the adjacent forest by stem base sprouting (Figure 2), below-ground rhizomatous growth, and branch layering (Figure 3) (Mallik 1993). Sprouting refers to vegetative reproduction from the base of the plant, rhizomatous growth refers to vegetative buds shooting off from underground rhizomes and layering refers to the vegetative growth on mature stems and branches in the peripheral part of the bush that fall on the ground and produce adventitious roots.

Thickets of ericaceous shrubs (Figure 4) inhibit canopy regeneration and influence forest structure (Phillips and Murdy 1985; Clinton and Boring 1994; Hedman and Van Lear 1995; Chastain and Townsend 2008; Beckage et al. 2000). Species richness as well as percent cover in the regeneration stratum found under ericaceous shrubs is reduced the denser a shrub thicket is, but even low densities of shrubs influence the regeneration layer (Baker and Van Lear 1998). Mortality of tree seedlings under shrub thickets tends to increase with age. Not only are tree seedlings reduced but the seed bank can also be lower in areas covered by shrub thickets compared with more open forests (Hille Ris Lambers and Clark 2003).



Figure 2: Multi-stemmed *Kalmia latifolia*. When collected this was considered a clonal group, photo by A. Pipkin, 2009.



Figure 3: Clonal reproduction of *Kalmia latifolia*. Vegetative layering and rhizomatous growth can be seen after removal of soil and duff, photo by A. Pipkin, 2009.





Figure 4: Ericaceous shrub thicket. This shrub thicket found in the Jefferson National Forest is primarily composed of *Kalmia latifolia* with sparse regeneration underneath, photo by A. Pipkin, 2009.

The reduction of light is thought to be the most important inhibitor of canopy regeneration (Beckage et. al. 2000; Clinton and Boring 1994), although many variables contribute to low regeneration levels under shrub thickets (Hille Ris Lambers and Clark 2003). Under a dense thicket of shrubs, light cannot penetrate to the forest floor to promote the regeneration of canopy species. After a canopy gap has been opened, if there are Ericaceae present, the understory will not receive enough light for canopy species to regenerate (Brose and Waldrop 2010). Carbon limitation in oak seedlings was found as a result of low light under shrub thickets. This limitation is another cause of early mortality in canopy seedlings (Beier et al. 2005).

The shrub layer also inhibits canopy regeneration and alters forest dynamics by accumulating nutrients, mainly nitrogen and phosphorus, in their evergreen leaves. Because it takes so long for their leaves to decompose the cycling of these nutrients is slow and has been cited as a cause for low canopy regeneration (Boring et al. 1981, Chastain et al 2006). Allelopathy of *Kalmia latifolia* and *Rhododendron maximum* has also been suggested as a potential inhibitor of the overstory (Eppard 2005; Nilsen et. al. 1999). Allelopathy is characterized by the chemical release of compounds from one plant to another inhibiting or stimulating growth (Rice 1984). If allelopathy does have any effect on tree establishment it is weak (Eppard et. al 2005), although non-significant reductions in performance of *Pinus rigida* were observed in a laboratory study.

The slowly decomposing layer of ericaceous leaf litter blanketing the ground seems to have a greater effect on regeneration than any allelopathic influences (Nilsen et. al. 1999). *Kalmia latifolia* leaves drop year around with a concentration in autumn

and spring and produce approximately  $127 \text{ kg ha}^{-1}$  in leaf litter biomass. This biomass is not easily decomposed because of the high fibers in the sclerophyllous leaves (Monk et al. 1985). These leaves are not consumed by herbivores because of their sclerophyllous properties (Chabot et al. 1982). Evergreen leaves can increase the build-up of duff and alter litter properties, making it more nutrient deficient than litter composed of deciduous leaves (Beckage et al. 2000). The shrub layer is not inhibited by its own thick duff layer initially, but it can be if seeds are the primary recruitment strategy, as opposed to sprouting. Removing the humus layer is necessary to promote suitable conditions for pine seedling establishment. It is possible to reduce the humus layer by hot fires or slow but high intensity smoldering combustion that consumes the humus as well as the underground plant parts (Mallik 2003).

#### Shrub Expansion and the Role of Disturbance

Anthropogenic impacts such as grazing, climate change, fire suppression and increases in atmospheric carbon have been cited as possible causes of shrub expansion around the world (Johnson 1990; Brown and Archer 1999; Mallik 2003; Hinman 2008). Recently there has been evidence of increases in the ericaceous shrub layer in the Appalachian Mountains (Phillips and Murdy 1985; Dobbs and Parker 2004). Factors such as fire suppression, cessation of grazing, logging and chestnut blight (McGee and Smith 1967; Phillips and Murdy 1985; Vandermast and Van Lear 2002) have been cited as possible causes of shrub expansion in the Appalachian region. Although there is not a consensus on why expansion is occurring, it has generally been attributed to changes in



the disturbance regime of the Appalachian Mountains and is likely the influence of multiple variables (Elliot et. al. 1999).

The disturbance regime in any forest ecosystem contributes to forest structure and composition. The historical legacy of disturbance in the southern and central Appalachian Mountains is important for understanding past vegetation patterns and how changes in disturbance history have led to current vegetation composition (Peters et al. 2006; Bolstad and Swank 1997). Humans have been altering the landscape of North America and the Appalachian Mountains for thousands of years (Delcourt and Delcourt 2008). In the last century European settlers have altered the disturbance regime of the southern and central Appalachians considerably. Beginning in the 1920s, establishment of protected national parks and national forests in this region created a different disturbance regime in xeric pine-oak stands. Protected areas resulted in a shift of disturbance frequency and type in the southern and central Appalachian Mountains (Pyle 1988).

Some studies evaluating the change in vegetation composition because of an alteration of the disturbance regime ignore the ericaceous shrub layer (Vandermast and Van Lear 2002). Yet changes in the disturbance regime may have led to an increase in the ericaceous shrub layer (Dobbs and Parker 2004). Pest invasions, strong winds, ice storms and disease frequently create canopy gaps (Brose and Waldrop 2010). The disturbance regime has converted to a system that commonly generates small canopy gaps surrounded by dense vegetation instead of more open canopy conditions caused by disturbances, such as fire. Smaller canopy opening do not promote vegetation recovery

as quickly as larger gaps (Shure et al. 2006). Ericaceous shrubs are able to occupy and outcompete canopy species in small gaps (Beckage and Clark 2000; Dobbs and Parker 2004). Once a canopy gap becomes available if the shrub layer is present it will utilize resources and inhibit the establishment of canopy species (Baker and Van Lear 1998; Clinton and Boring 1994; Hille Ris Lambers and Clark 2003; Vandermast and Van Lear 2002).

Disturbance type is important when considering their effects on the ericaceous shrub layer. While fire may reduce shrub cover, disturbances such as ice storms and SPB could potentially benefit the shrub layer (Lafon and Kutac 2003). The gaps created by disturbance events such as ice storms and SPB outbreaks are not large enough or frequent enough to promote yellow pine regeneration with the presence of ericaceous shrubs that remain mostly intact after these disturbance events (Beckage et al. 2000).

Disturbances that remove the canopy without removing the shrub layer may perpetuate the existence and expansion of ericaceous shrubs. Although some canopy species such as *Acer rubrum* and *Tsuga canadensis* may be able to regenerate under dense shrub thickets it appears they are not, and Ericaceae seem to be self-replacing and spreading into areas in which they previously may have been scarce (Baker and Van Lear 1998). Canopy gaps caused by small infrequent disturbances are considered drivers of diversity and heterogeneity in temperate forests. The spatial and temporal scale of disturbance events contributes to heterogeneity on the landscape (Clinton and Baker 2000). The presence of ericaceous shrubs has more impact on overstory seedling regeneration, species richness and diversity than disturbance-related canopy gaps

(Beckage et al. 2000). Although it has been asserted that canopy gaps caused by disturbances promote heterogeneity on the landscape, the occupation of these small gaps by ericaceous shrubs reduces their importance as drivers of diversity and landscape heterogeneity (Clinton 2003; Clinton and Boring 1994; Beckage et. al. 2000).

After a disturbance, whatever species can occupy an area first tends to be the most likely to establish (Noble and Slayter 1980). The most successful species are often sprouters, because they already have an extensive underground system that allows them to quickly occupy an area. Sprouting can favor self-replacement, altering forest dynamics, reducing the importance of recruitment and emphasizing the role of persistence (Bond and Midgely 2001; Dietze and Clark 2008). *Kalmia latifolia* can sprout from damaged adult stems and roots after a disturbance while yellow pines in the Appalachian Mountains regenerate primarily from seeds with very limited sprouting capabilities (McGee and Smith 1967; Harrod et. al. 2000). Brose and Waldrop (2010) found that where *Kalmia latifolia* was present even oak, a species that can also regenerate vegetatively did not. The result of prolific sprouting by shade-tolerant Ericaceae means that once high densities establish it may become difficult to restore a system back to low shrub cover. Once established ericaceous shrubs will persist on the landscape and gradually begin to advance into new territory (Baker and Van Lear 1998; Dobbs and Parker 2004).

Shrub dynamics of Ericaceae in the Appalachian Mountains are disputed. A study using aerial photography to analyze shrub dynamics in the Southern Appalachian Mountains found that the ericaceous shrub layer expanded 8% from 1976 to 1993 in an

area approximately 1,653 ha. This occurred at the same rate in both an unmanaged and an anthropogenically disturbed watershed in the Appalachian Mountains (Dobbs and Parker 2004). However, when analyzing the understory using aerial photography it can be difficult to characterize the understory composition of the pixels. After clear-cutting in an Appalachian forest, the density of both *Kalmia latifolia* and *Rhododendron maximum* stems increased rapidly, but biomass increases were slow (Boring et al. 1981). McGee and Smith (1967) found that shrub thickets were not expanding as long as they were left undisturbed.

Management of this diverse forest system needs to be re-evaluated to preserve xeric pine-oak communities with Appalachian Mountain endemics such as *Pinus rigida*. If forests become overrun with shrub thickets, their aesthetic value, recreational value, timber resources and wildlife suitability would be degraded and may be difficult to restore. In some cases only stand-replacing fires, clear-cuts or herbicidal treatments may return the forest to a low shrub cover state (Brose and Waldrop 2010). Because of high costs, mechanical and herbicidal removal may not be viable options (Hooper 1969). Shrub management has important implications not only for natural recreation areas in the southern Appalachian Mountains but also timber industries working in national forests. If ericaceous shrubs inhibit growth of valuable timber species, national forests stand to lose important economic resources (Hooper 1969).

In the following sections I will outline the predominant biotic and abiotic disturbances of the central and southern Appalachian Mountains. These sections briefly

discuss how the disturbances affect vegetation, in particular ericaceous shrubs, in the Appalachian Mountains.

### Biotic Disturbances

In the early twentieth century American chestnut (*Castanea dentata* (Marsh.) Borkh.) was a dominant tree species in the Appalachian Mountains (Braun 2001). Chestnut tree blight fungus (*Cryphonectria parasitica* (Murrill) M.E. Barr), which causes mortality in *Castanea dentata*, was first found in the United States around the 1900s. Symptoms of blight include bark cankers, wilting foliage and epicormic sprouting below the cankers (Anagnostakis 1987). *Cryphonectria parasitica* removed up to 99% of *Castanea dentata* by the 1930s. This caused major vegetation shifts throughout the eastern United States, such as opening the canopy, replacement by understory trees and shrubs, changes in the humus and advancement of subordinate species (Keever 1953). After *Castanea dentata* was removed *Rhododendron maximum* and *Kalmia latifolia* increased in its place (Day and Monk 1974). Vandermast and Van Lear (2002) suggested that *Rhododendron spp.* may have replaced *Castanea dentata* as the keystone dominant in the southern Appalachian Mountains.

The southern pine beetle (*Dendroctonus frontalis* Zimmermann) is a native species of the southeastern United States. Southern pine beetle (SPB) primarily affects yellow pine species. An outbreak can kill a few pines in a small area or large swaths of pines covering several hundred acres. (Thatcher et al. 1982). SPB has reduced the pine overstory on more xeric sites and preferentially infests older larger yellow pines (Knebel and Wentworth 2007; Lafon et al. 2007). Canopy gaps caused by SPB may have

allowed *Kalmia latifolia* to increase on these sites by taking advantage of resources, including light and nutrients. In dense fire suppressed pine sites in the Appalachian Mountains there have been increases in SPB infestation over the past century (Knebel and Wentworth 2007). Fire builds resistance to infestation by reducing stand density and increasing resin production of pines, making them more resistant to SPB outbreaks (Knebel and Wentworth 2007). Historically fire along with SPB kept stands stable (Lafon et al. 2007).

The exotic pest hemlock wooly adelgid (*Adelges tsugae* Annand) has recently infested hemlocks (*Tsuga canadensis* (L.) Carrière) in the Appalachian Mountains. The hemlock wooly adelgid is an exotic aphid that was introduced from Asia to the eastern United States in the 1940s and began to disperse north and south. The dispersal of this insect seems to be influenced by wind, mammals and logging, but migrating birds may be responsible for the most spatially extensive spread (Orwig and Foster 1998). It is probable that the combined result of removing sap and injecting toxic spittle in the tree causes needle loss and bud mortality, eventually resulting in branch and tree mortality within about four years (McClure 1987). The only canopy species that does not seem inhibited from growing under shrub thickets is *Tsuga canadensis* (Phillips and Murdy 1985; Vandermast and VanLear 2002). Because of hemlock wooly adelgid infestation, *Tsuga canadensis* is no longer a viable competitor, meaning that canopy regeneration under the dense shrub thickets may be even more limited. High shrub establishment could occur in stands affected by hemlock wooly adelgid, because of the high stability,

moist soils and infrequency of fire and other disturbances in these stands (Orwig and Foster 1998).

The Gypsy Moth (*Lymantria dispar* L.) was introduced to Massachusetts, USA from Europe in the late 1860s where it increased very rapidly. The destructive stage of the moth is during the larval period. Gypsy moth is not selective in the species of trees it defoliates. A total of 485 different species have been recorded as hosts (Forbush 1896). Gypsy moth outbreaks have increased *Kalmia latifolia* in stands dominated by hardwoods (Chastain and Townsend 2008). If *Kalmia latifolia* is present in the understory during a gypsy moth outbreak then it may impede forest recovery from this disturbance. The combination of increases of deciduous species in xeric sites and gypsy moth outbreaks could lead to expansion of *Kalmia latifolia*.

#### Abiotic Disturbances

Following European settlement, logging removed much of the timber and degraded the forests and soils of the Appalachian Mountains. Large tracts were logged for corporate timber as well as human settlement, farming and ranching (Pyle 1988). Before 1880 logging in this region was done by farm families clearing fields and building homes. After 1880 commercial logging came into the area and until about 1900 this meant the areas that were easily accessible in the rugged mountainous terrain were the only areas affected by clear-cutting. During the 1900s multiple timber agencies bought vast tracts of forest. Skidders and other industrial logging equipment became common tools for transporting timber to railroads from high elevations. Railroads became widely accessible for the transport of timber in more remote areas.

Technological innovations and conveniences ended selective clear-cutting and increased timber extraction, resulting in major alterations to the landscape (Lambert 1961).

Logging still occurs in this region but not at the same intensity as before the 1920s. In logged stands species with the primary regenerative strategy of sprouting tend to re-occupy the area first (Shure et al. 2006). For example a black spruce forest in Newfoundland was converted to a shrub heathland after logging (Mallik 1987). Removal of forest cover because of logging can cause major changes where Ericaceae are abundant, which may eventually lead to areas dominated by shrub heathlands.

Rare but influential major wind events usually caused by hurricanes and sometimes intense thunderstorms occasionally occur in the Appalachian Mountains (Elliott et. al. 2002). Wind events create canopy gaps by snapping or uprooting trees, subsequently resulting in the formation of pit and mound topography in a localized area. This causes changes in light, moisture, nutrients and vegetative cover. The trees that form pit and mounds are primarily canopy trees, although smaller understory trees are usually affected indirectly from the impact of a falling overstory tree. Indirectly damaged trees in the understory tend to have more sprouts than directly affected overstory trees (Clinton and Baker 2000). Because ericaceous shrubs are more likely to be indirectly affected this allows the understory layer to contribute extensively to vegetation recovery by occupying gaps quickly.

Ice storms also influence Appalachian Mountain Forests by reducing cover and creating gaps (Whitney 1984). However on sites dominated by yellow pine species it seems this disturbance causes higher mortality among yellow pines allowing shrubs and



hardwoods to occupy the gaps that are created by the disturbance (Lafon and Kutac 2003).

### Fire Disturbance History

Fire is a unique disturbance that consumes both dead and living biomass. Fire can alter both the environment and the vegetation it effects (Bond and Keely 2005). Fires occurred on the Appalachian landscape well before European settlement. In most of the eastern U.S. lightning-caused fires are relatively rare (Abrams and Nowacki 2008). The majority of pre-settlement fire events likely were caused by Native Americans. These burns were influential on the current forest composition. Before European settlement of the Appalachian Mountains, Native Americans burned frequently to attract game animals, prepare planting sites and increase oak species for acorn harvesting. Burning by Native Americans may have encouraged sprouters like oaks and chestnuts, and maintained ridgetop species, such as *Pinus pungens* L., that require fire (Delcourt and Delcourt 1997; Brose et. al. 2001). Fires usually occurred in autumn or early spring, but occasionally also occurred during the growing season due to drought (Aldrich 2011) or as a result of lightning (VanLear and Waldrop 1989).

While land clearing practices resembled those of Europe, fires used for hunting, herding, farming and logging all resembled practices used by Native Americans (Pyne 1997). Settlers continued to use fire until the acquisition of federal lands and the period of intense logging (Pyle 1988). Large fires caused by the logging industry started occurring around the mid-1800s and early 1900s. Catastrophic fires were caused by the combination of leaving large slash piles behind after logging and the use of industrial

logging equipment creating ignition sources. Fires were more severe during periods of drought (Lambert 1961; Pyne 1997). The catastrophic fires that occurred as a result of the logging industry and combined with drought burned large areas and caused fatalities, contributing to negative public perceptions of fire (Pyne 1997; Brose et al. 2001 ). After the Weeks Act passed in 1911 national forestland was acquired in the eastern United States. One of the first priorities of the forest service was fire suppression, in the national forest (Brose et al. 2001 ). Since around the 1930s and 1940s the fire management strategy in the Appalachian Mountains has been to suppress and prevent all fires. Fire suppression has caused a major alteration in fire frequency clearly shown in the dendroecological record that extends back to the 1600s (Harmon 1982; Shumway et al. 2001; DeWeese 2007; Hoss et al. 2008; Aldrich et. al 2010).

#### Fire Suppression and Vegetation Response

It has been asserted that fire is a dominant control in maintaining oak-pine forests in the Appalachian Mountains and across the eastern United States (Harmon 1982; Abrams 1992; Barden 2000; Harrod 2000; Waldrop 2000; Brose et al. 2001 ; Shumway 2001; DeWeese 2007; Lafon 2007; Abrams and Nowacki 2008; Aldrich 2011). The vegetation in oak-pine communities are shade-intolerant and require large canopy gap openings that occur with frequent burning (Lafon 2007). Yellow pine species found on these sites are considered seral species in which high disturbance frequencies are required to maintain their presence on the landscape. Only the poorest sites in the Appalachian region can maintain yellow pine species without frequent disturbance (Barden 1976).

Fire suppression has resulted in increases in fire-intolerant species, such as *Acer rubrum* and *Pinus strobus* in forests previously dominated by yellow pines and *Quercus spp.* (Schowalter 1981; Harrod et.al. 1998; Aldrich et. al. 2010). Fire suppressed pine forests have increased stand density as well as encroachment of hardwoods and *Kalmia latifolia* (Harrod et. al. 1998; Waldrop 2000; Lafon et al. 2005). The increase in fire sensitive species has been well documented while the possibility of shrub layer expansion is sometimes ignored, although the ericaceous shrub layer may be replacing pine stands affected by fire suppression and pest disturbance at an equal or greater rate than hardwood species (Brose and Waldrop 2010). Because of fire suppression on these landscapes fire resistant stands have become more dense (Lafon et al. 2007) causing there to be less resources and making these stands structurally less resilient to disturbances (Doren 1993).

The frequent burning that occurred before fire suppression would have halted woody fuel accumulation (Aldrich et al. 2010). It is crucial to inhibit the creation of large fuel loads. The fuel load that accumulates due to infrequent and/or low intensity prescribed burning allows severe fires to occur, destroying the canopy (Waldrop et al. 2007). Fuel loads have not only accumulated due to a higher density of shrubs, but pest disturbances have allowed the accumulation of decomposing woody debris (Waldrop et al. 2007). A high amount of humus can develop from ericaceous shrub leaf litter. This creates a duff layer that can be difficult to ignite reducing the possibility of fire as an effective mechanism to remove it. At a certain point, the duff layer could accumulate to such high amounts that low intensity fire may not be able to burn and only high intensity

smoldering burns would be successful in reducing the humus layer (Mallik 2003). Post-fire litter depth has a positive correlation with density of ericaceous shrubs, this implies that if litter is not removed after fire then shrub cover will likely remain high (Harrod et al. 2000), further indicating that milder fires are not effective in removing the shrub layer. Fires must be intense enough to burn through the humus layer, open serotinous cones and reduce ericaceous shrub cover for pine regeneration to occur (Elliot et al. 1999).

Frequent fires have been shown to reduce cover of the evergreen understory layer, maintaining more open stands (Clinton and Boring 1994; Elliot et al. 1999; Harrod et al. 2000). Before the era of fire suppression surface fires that top-killed hardwood trees and shrubs were common, but occasional crown fires would also occur, creating age cohorts of pines (Brose et al. 2001; Aldrich et al. 2010). High intensity fires on xeric sites reduce the abundance, density and basal area of *Kalmia latifolia* (Clinton and Boring 1994). Low intensity surface fires in this region would not cause canopy mortality but would often reduce the understory and fire sensitive species including ericaceous shrubs (Harrod et. al. 1998; Waldrop et al. 2000). Reduction in understory density results in increases of herbaceous cover and diversity after a fire event (Arthur et al. 1998). Fire suppression has led to positive feedbacks encouraging higher percentages of shrub cover (Dobbs and Parker 2004) and decreased recruitment of pines in xeric oak-pine communities (Clinton et al. 1993; Waldrop et al. 2000). Although the evergreen shrub layer is reduced after prescribed burns it may rebound due to resprouting if frequency of fire is too low (Elliott et al. 1999). Winter fires are the least effective at

reducing the shrub layer (Harrod et al. 2000). Fire characteristics such as seasonality, frequency and intensity all influence shrub establishment in xeric pine-oak communities.

Studies measuring abundance of ericaceous shrubs after fire claim that although sprouting occurs, fire causes ericaceous shrubs to no longer be viable competitors.

Because of the reduced stature of ericaceous shrubs in this environment canopy species can regenerate successfully (Clinton et al. 1993; Baker and Van Lear 1998; Arthur et al. 1998; Ducey et al. 1996). Arthur et al. (1998) found that after a single fire the shrub layer increased but after two fires the shrub layer decreased in density below unburned stands, this study measured stems per hectare and not basal area. A study by Ducey et al. (1996) found that 8 years after fire unburned plots had a lower density of *Kalmia latifolia* than burned plots but that the difference is greater after a moderate burn compared to an intense burn. While unburned plots had irregular patches of large *Kalmia latifolia*, moderately burned plots had a continuous cover of small statured individuals with few gaps (Ducey et al. 1996). The continuous layer did not seem to reduce diversity or establishment at these sites.

McGee and Smith (1967) found shrub thickets are not expanding and management of these forests could be postponed, yet they offer no management strategy. Other research suggests as the time since the last fire event increases shrub cover will rebound from any reduction caused by fire (Holzmueller 2009). There are a number of reasons the influence of fire on the shrub layer may differ. Differences in response to fire can occur when the pre-fire composition and structure of an area differs as well as

the fuel loads, severity of the fire event and season in which the fire occurred (Harrod et al. 2000).

Previous research has shown that the increase in the ericaceous shrub layer inhibits canopy regeneration. What is not as well understood is the behavior of the shrub layer during the last century and how it has been influenced by fire suppression. Before fire suppression there was probably an interaction between fire and landscape pattern (Peterson 2002). This pattern may no longer be discernable and may no longer influence fire patterns. It seems likely that the historic fire regime inhibited the spread and reduced the abundance of ericaceous shrubs. Other studies suggest that this may have encouraged the growth of yellow pine and other fire dependent xeric species (DeWeese 2007; Aldrich 2011). I hypothesize that there have been increases in ericaceous shrubs because of an alteration in disturbance regime. Because there has been very little fire on this landscape for the past century nobody knows with any certainty if fire will have the desired effect on vegetation dynamics in the Appalachian Mountains. Although the dendroecological record suggests that the Appalachian Mountains had frequent fires this re-introduction may or may not return vegetation to its pre-fire suppression composition (Harmon 1984).

## CHAPTER III

### METHODS

#### Study Area

Four sites that stretch across the southern and central Appalachian Mountains were used for the study area (Figure 5). Goldmine Trail is the most southern site, located in the Tennessee portion of Great Smoky Mountains National Park. This site is a part of the Blue Ridge Province of the Appalachian Mountains. Griffith Knob and Brush Mountain are farther north, located in the Jefferson National Forest. The most northern site, Reddish Knob, is located in the George Washington National Forest. These three northern sites are located in the Ridge and Valley province of the Appalachian Mountains.

#### Vegetation

The forests of the eastern half of the United States are considered eastern deciduous forest because of the deciduous tendency of most of the vegetation found in this region, however, these forests are complex and contain many evergreen species both in the arboreal and shrub layer (Braun 2001). The Appalachian Mountain forests are the highest, wettest and most densely vegetated in the eastern deciduous forest (Bowman 1911). Forests in this region are typically considered multi-storied with an overstory, mid-story and herbaceous layer (Delcourt and Delcourt 2000). Basins, gorges and coves, along with differences in altitude, aspect and moisture, have led to high species diversity (Bailey 1980). High relief from valleys to ridgetops result in the distribution of different elevational zones of vegetation (Braun 2001). Although major shifts in climate,

vegetation distribution and ranges have occurred there have been no glaciations within the southern region of the Appalachian Mountains allowing the occupation and evolution of flora and fauna here for many millions of years, which contributes to the regions high diversity (Whittaker, 1956).

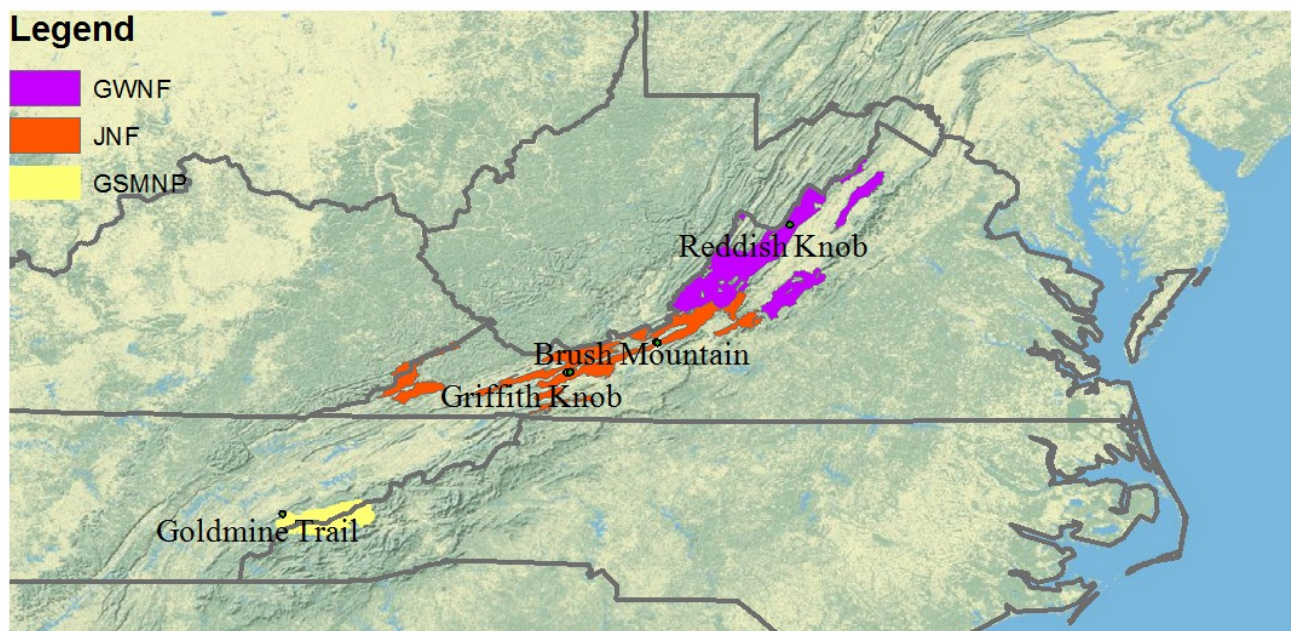


Figure 5: Study Area. This research was conducted in the George Washington National Forest, Jefferson National Forest and Great Smoky National Park, located in the central and southern Appalachian Mountains.



There are many different forest classifications for the Appalachian Mountains. Bailey (1980) classifies the Appalachian Mountain forest as the Central Appalachian Broadleaf Forest. More specifically the southern and central Appalachian Mountains are characterized as the Oak-Chestnut Forest Region (Braun 2001), although the Oak-Chestnut association in this region actually no longer exists after the demise of American chestnut (*Castanea dentata*). During the 1920s, chestnut blight (*Cryphonectria parasitica*) removed nearly all *Castanea dentata* from the landscape (Bailey 1980). A more contemporary classification would be an oak-hickory forest with primary species including northern red oak (*Quercus rubra* L.), chestnut oak (*Quercus prinus* L.), white oak (*Quercus alba* L.) and tuliptree (*Liriodendron tulipifera* L.) (Keever 1953; Braun 2001). This forest type is limited to the mountainous areas of the Appalachian Mountains (Braun 2001). These forest classifications are generalized over the entire Appalachian Mountains.

In addition to generalized classifications of Appalachian Mountain forests there have been more specific classifications of the region. Whittaker's (1956) study of the southern Appalachian Mountains defined four communities for which tree species are assigned based on moisture gradients, and 15 sub communities in which the driving controls are moisture, exposure and elevation. There is usually a subtle transition in flora between these communities. The first of these four basic vegetation community types are mesic communities, or communities with abundant moisture. Cove forests, because of their natural beauty, are the most commonly recognized mesic community in the Appalachian Mountains. Cove forests include a dense canopy with smaller trees in

the understory and an abundance of ferns and herbs in the summer. Eastern Hemlock (*Tsuga canadensis*) forests are also considered mesic and usually occur in shaded ravines. Grey Beech (*Fagus grandifolia*) forests are the third mesic forest type that occur on both north and south slopes and are characterized as open undisturbed sites. The shrub layer used to be almost absent from mesic sites, however, with the loss of *Castanea dentata* and *Tsuga canadensis* there has been an increase in *Rhododendron* spp. occurring in mesic forests, especially near streams (Clinton and Boring 1994; Baker and Van Lear 1998).

Sub-mesic plant communities found in the Appalachian Mountains are slightly drier than mesic communities. The third major community type, sub-xeric, is dryer than both mesic and sub-mesic sites. The forests in these two major community types usually consist of oak-chestnut and oak-hickory forests. Chestnut oak-chestnut forests and red oak-chestnut forests are considered sub-mesic, with common species including *Acer rubrum*, *Carya tomentosa* and *Quercus rubra*. The white oak-chestnut forest and oak chestnut heaths are considered sub-xeric with common species including *Quercus alba* and *Oxydendrum arboreum*. There will be more *Pinus* spp. found in the sub-xeric sites than the sub-mesic sites (Whittaker, 1956).

The most moisture deficient vegetation group is the xeric community, composed of pine stands, along with a few *Quercus* spp. such as *Quercus montana* and *Quercus coccinea*. These stands occupy slopes with a south-to-southwest facing orientation (Whittaker 1956). In the early twentieth century these stands were dominated by early-successional communities that had relatively open canopies allowing abundant light to

penetrate to the forest floor. Because of the absence of fire needed to maintain this forest type stand density has increased (Harrod et. al, 2000). The three xeric pine stands Whittaker describes are virginia pine forest, pitch pine heaths and table mountain pine heaths. Virginia pine (*Pinus virginiana* Mill.) stands are found at low elevations below 900 meters most often with *Quercus coccinea* (Münchh). Shrub coverage usually ranges from 10% to 40%. In the xeric and sub-xeric sites the dominant shrub found is *Kalmia latifolia*. Historically shrub cover has increased toward xeric sites, with a decline in the wettest mesic communities and a peak in diversity in sub-xeric communities. Pitch pine (*Pinus rigida* Mill.) stands can be found at middle elevations of 650 to 1,000 meters. Table mountain pine (*Pinus pungens* Lamb.) stands are found at higher elevations (Whittaker 1956). *Pinus pungens* and *Pinus rigida* stands are usually found between southern and western facing slopes (Brose and Waldrop 2006) and are adapted to fire, have thick bark and serotinous cones (Aldrich 2011). These stands are adapted to a higher frequency burning regime than *Pinus virginiana* stands (Zobel 1969). Shrub coverage can be high in the xeric pine forest where light penetration can reach the forest floor, but in the most xeric stands shrub coverage may be low. Historically diversity in *Pinus pungens* stands has been low because the species is usually so dominant (Whittaker 1956). Today, however, many xeric pine stands contain fire-intolerant species in the understory such as *Acer rubrum* and *Nyssa sylvatica* which are often found in more mesic communities (Brose et al. 2001 ; Aldrich et al. 2010). Xeric pine communities across the Appalachian Mountains have been reduced in extent over the past century due to fire exclusion (Harrod et. al, 1998). Pines are an important

component in maintaining high landscape diversity of the Appalachian Mountains, and their reduction is of concern to land managers.

The three most northern sites of the study area are located in mixed table mountain, pitch pine heaths (Figure 6). Goldmine Trail is located in a virginia pine and Shortleaf Pine forest and is warmer and wetter than the other three sites. The Goldmine site has a more gentle slope than the other three sites and is different from the other three sites because of elevation and vegetation composition. Shrub coverage varied across plots and sites with some plots having little to no shrubs (Figure 7) and others having very dense coverage (Figure 8).

#### Geology and Physiography

The Appalachian Mountains are an old, relatively low lying mountain chain in eastern North America. They extend from Alabama to Newfoundland, around 3,000 km (Rodgers 1970). The Paleozoic was the last major era of construction of the Appalachian province, since this time diminution has been occurring (Kentucky Geological survey). Relatively shallow soils blanket the area (Bowman 1911). These mountains are characterized as subdued with folded strata (Bailey 1980). Geographically these mountains are not a single continuous range but are actually a group of mountains that run roughly parallel to each other. This mountain chain reaches its highest point in North Carolina on top of Mt Mitchell at 2,037 meters. The last ice age covered all of the northern Appalachian Mountains leaving everything below New York unglaciated although periglacial phenomena have been found as far south as North Carolina (Rodgers 1970).



Figure 6. Xeric pine forest. A Pine Heath Community near Reddish Knob in the George Washington National Forest. The lighter green canopy is deciduous forest that surrounds the darker green coniferous stands, composed of *Pinus pungens* and *Pinus rigida*, photo by A. Pipkin, 2009.





Figure 7: Sparse ericaceous shrub cover. This sparsely populated plot at Griffith Knob had pine regeneration, photo by A. Pipkin, 2009.



Figure 8: Dense ericaceous shrub cover. High density plot, at Reddish Knob, before the removal of shrubs, photo by A. Pipkin, 2009.

Geologically the Ridge and Valley is underlain by sedimentary rock that has been heavily folded and faulted but not metamorphosed, except for cleavage in the shale and limestone along the southeast side. This region represents the eastern margin of the Paleozoic interior sea (Rodgers 1970). The sedimentary rocks formed large sheets that are stratified. Geologically the Blue Ridge is composed of mostly crystalline rocks that are some of the oldest in the province (Kentucky Geological Survey). This area is primarily composed of schists, gneisses and granite with crystalline rock metamorphosed during the Paleozoic mountain building period (Bowman 1911). These rocks thrust from under weaker rocks of the Ridge and Valley province (Rodgers 1970). The crystalline structure of these rocks is much more resistant than the Triassic sediments beneath the piedmont region to the east (Fenneman 1938) and likely contributed to the formation of the sedimentary rocks of the ridge and valley to the west (Kentucky Geological Survey).

The Ridge and Valley Physiographic province is west of the Blue Ridge Province and is characterized by narrow ridges that persist for hundreds of kilometers, parallel to one another in the direction of the mountain chain. The Ridge and Valley extends with few interruptions from one end to the other. The drainage pattern found here in earlier periods was transverse but longitudinal streams later developed making a trellised drainage pattern with long streams running through the parallel valleys (Fenneman 1938).

The Blue Ridge Province is characterized by a single massive ridge. The Blue Ridge Province lies west of the piedmont province that borders the coastal plain of the eastern United States. The complicated structure of mostly crystalline with some

igneous and metamorphic rock has formed a complex dendritic drainage pattern. This section of the Appalachian Mountains is the most rugged, but like the rest of the Appalachian Mountains retains a smoothed sloping appearance (Bowman 1911). Craggs, bare cliffs and un-vegetated talus slopes are rare with most summits being rounded and occupied by forests. The Blue Ridge province also has many coves in which a flat fertile limestone surface is surrounded by mountains. The southern half of the Blue Ridge province is the widest at around 100 km across (Fenneman 1938).

### Climate

The climate of this area is characterized as temperate (Bailey 1980). The entire mountain chain is subject to both maritime air masses brought in from the Gulf of Mexico and the Atlantic Ocean and continental air masses brought by the westerlies, which is the dominant wind pattern for this latitude. Occasionally tropical storms and hurricanes make their way into the Appalachians bringing high winds and heavy rains (Whiteman 2000).

Because of elevation the amount of cloud cover and rainfall are greater in the mountains than the surrounding lowlands. Topography causes differences in the distribution of climatic conditions throughout the area. Southeastern slopes that directly face the sun for several summer months receive almost an equivalent amount of sun as equatorial areas, causing them to be much warmer and drier than the northwestern slopes (Bowman 1911). Mountain ranges cause orographic uplift carrying air masses up the windward side of the mountain causing rain to fall on this side with little moisture remaining when the air mass reaches the leeward side (Whiteman 2000). The southern



and central Appalachian Mountains receive the most rainfall in the spring, with evenly distributed rainfall for the remainder of the seasons. The median yearly temperature, maximum and minimum temperatures and mean annual precipitation describe the climate of these four sites (Table 1). The median yearly temperature is computed by finding the median of the mean yearly temperature values from 1971-2000. The maximum and minimum temperatures are the maximum and minimum yearly mean values for temperature from 1971-2000.

Table 1: Climate of study area. Climate Station data that corresponds with each site, data taken from NCDC and the Southeast Regional Climate Center.

<b>Site</b>	<b>Climate Station</b>	<b>Elevation (m)</b>	<b>Median Yearly Temp (°C)</b>	<b>Max Temp(°C)</b>	<b>Min Temp(°C)</b>	<b>Mean Annual Precip (cm)</b>
Goldmine	GATLINBURG	443	12.9	25	-4.9	145.2
Griffith	WYTHEVILLE	747	10.5	23.7	-7.7	96.1
Brush	BLACKSBURG	640	10.6	24.1	-7.3	108.3
Reddish	HOT SPRINGS	682	10.6	23.7	-8	109

## Fire

A study of fire in three physiographic provinces of the Appalachian Mountains from 1970-2003 shows anthropogenic and natural fires are higher in the Blue Ridge than the Ridge and Valley (Lafon et al. 2007). Fires usually occur during the spring and fall which correspond with high fuel accumulation and dry windy weather. On average the Blue Ridge has smaller anthropogenic fires than the Ridge and Valley. This difference may be related to influences of fire spread and landscape structure. In general the Blue Ridge is more fire prone than the Ridge and Valley.

Dendroecological reconstructions of fire history have been prepared for my study area (Deweese 2007; Aldrich 2011; Laforest unpublished). On all the sites the last widespread fire event occurred between 1910-1930 (Figure 9). This abrupt change in the fire regime after the 1930s, due to fire suppression practices on public lands, is the only large scale shift in the fire regime since the 1600s according to the dendroecological record.

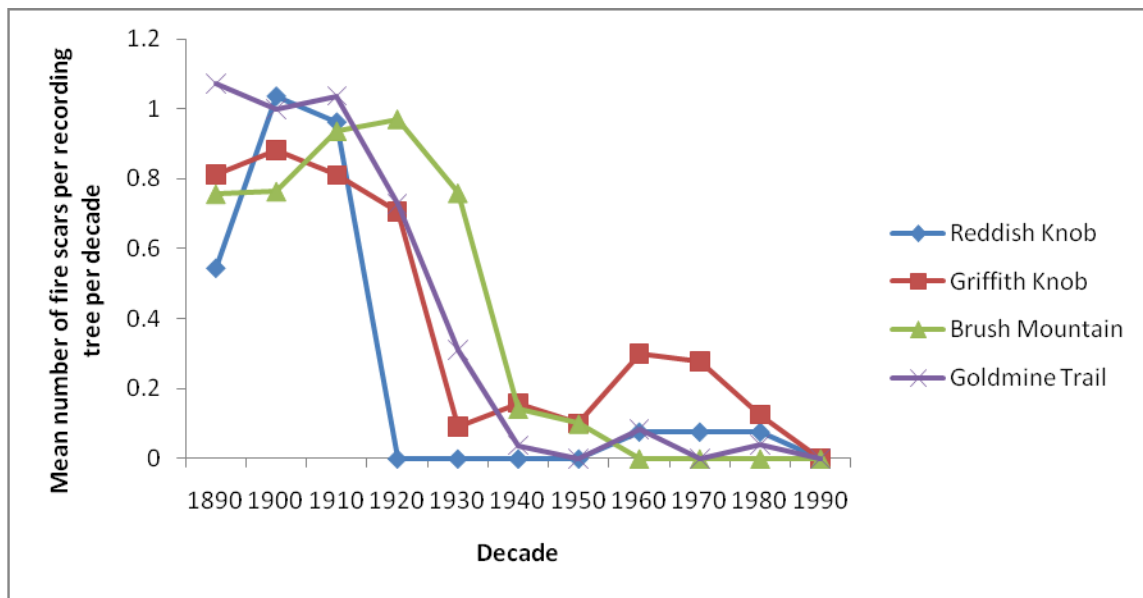


Figure 9. Fire frequency per decade. Temporal trend of fire events per decade on all four sites determined from dendroecological data (DeWeese 2007; Aldrich 2011; LaForest Unpublished). The chart shows the mean number of fire scars for every recording tree for each decade. The drop in the early 20<sup>th</sup> century of fire activity began at Reddish Knob. There is a small renewal of fire events on Griffith Knob between the 1960s and 1970s. Before 1890 there is a long history of frequent fires on these sites extending back to the 1600s or 1700s.

The last widespread fire that affected Goldmine Trail was in 1929, but the most recent fire event was in 1982. Griffith Knob has the most frequent occurrences of fire over the past century. Brush Mountain had a widespread fire in 1926 and another large event in 1934. The 1934 fire is the most recent large fire that has occurred on any of the sites sampled. The year 1957 marked the last fire shown in the dendroecological record. A widespread fire occurred in 1913 on Reddish Knob. This is the longest period of time that any site has been without a widespread fire. There have been four small isolated fire events on Reddish Knob since this time between 1958 and 1985. The fire scars for these events were only found on a single tree.

Griffith Knob has had the most frequent burning during fire suppression, because of a resurgence of fire in the 1960s. Reddish Knob has had the least amount of burning in frequency, extent and intensity. Brush Mountain, like Reddish Knob, has had very little burning during the last 90 years, but the last major fire on Brush Mountain occurred about 20 years after the last major fire at Reddish Knob.

#### Southern Pine Beetle

The southern pine beetle is a major biological disturbance that occurs in pine forest in the Southeast (Schowalter et al 1981). The record of SPB outbreaks acquired from the United States Forest Service begins in 1960 and ends in 2000 for Tennessee and 2001 for Virginia (Figure 10). The most southern site Goldmine Trail is located in Blount County, TN which has been affected by SPB 11 times since 1960 (Forest Health Atlas). The field observations showed clear evidence of a recent SPB outbreak at Goldmine Trail. There were many dead pines that were fallen and a few pine snags still standing as a result of SPB. The Virginia sites did not have clear evidence of SPB outbreaks while data were being collected, although both Reddish Knob and Brush Mountain were in counties affected by SPB during the 1990s.

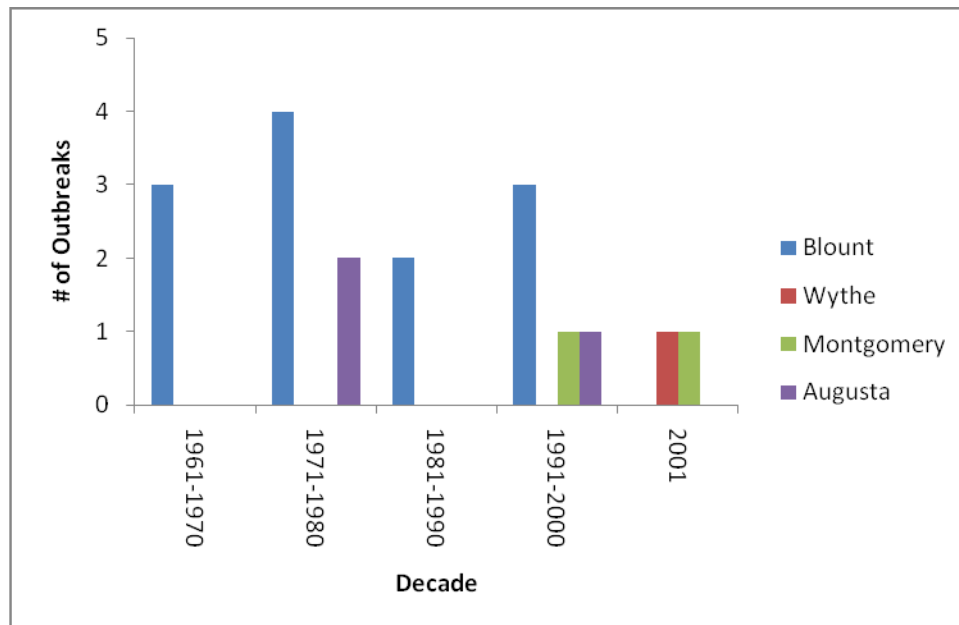


Figure 10: Southern Pine Beetle outbreaks. The frequency of Southern Pine Beetle Outbreaks for each decade for the county where each site is located (USDA, Forest Health Atlas). There is no data for the last decade on Blount County and only one year of data for the other counties.

### Field Methods

The fieldwork for this research took place in the summer of 2009. The data were collected beginning with Goldmine Trail in May. A second trip was taken in July, when data were collected at the Virginia sites. Two transects were established at each of the four study sites on different slopes. These slopes were chosen because fire histories had previously been reconstructed there. At each site we randomly chose the ridgetop plot of each transect. From each ridgetop plot we ran a measuring tape down the entire length of the slope. After establishing the length of the slope, it was divided so that there were five plots evenly spaced along the length of the entire transect (Table 2). Each plot was

2.5 m perpendicular to the slope and 5 m parallel to the slope of the mountain. The five plots included a ridgetop plot, an upper mid-slope plot, a midslope plot, a lower mid-slope plot and a slope bottom plot (Figure 11).

Table 2: Length of transects. The distance (m) from the ridgetop to the slope bottom of each transect. Each site had two transects.

Site	Goldmine Trail	Griffith Knob	Brush Mountain	Reddish Knob
Transect 1	70	156	115	164
Transect 2	105	253	110	178

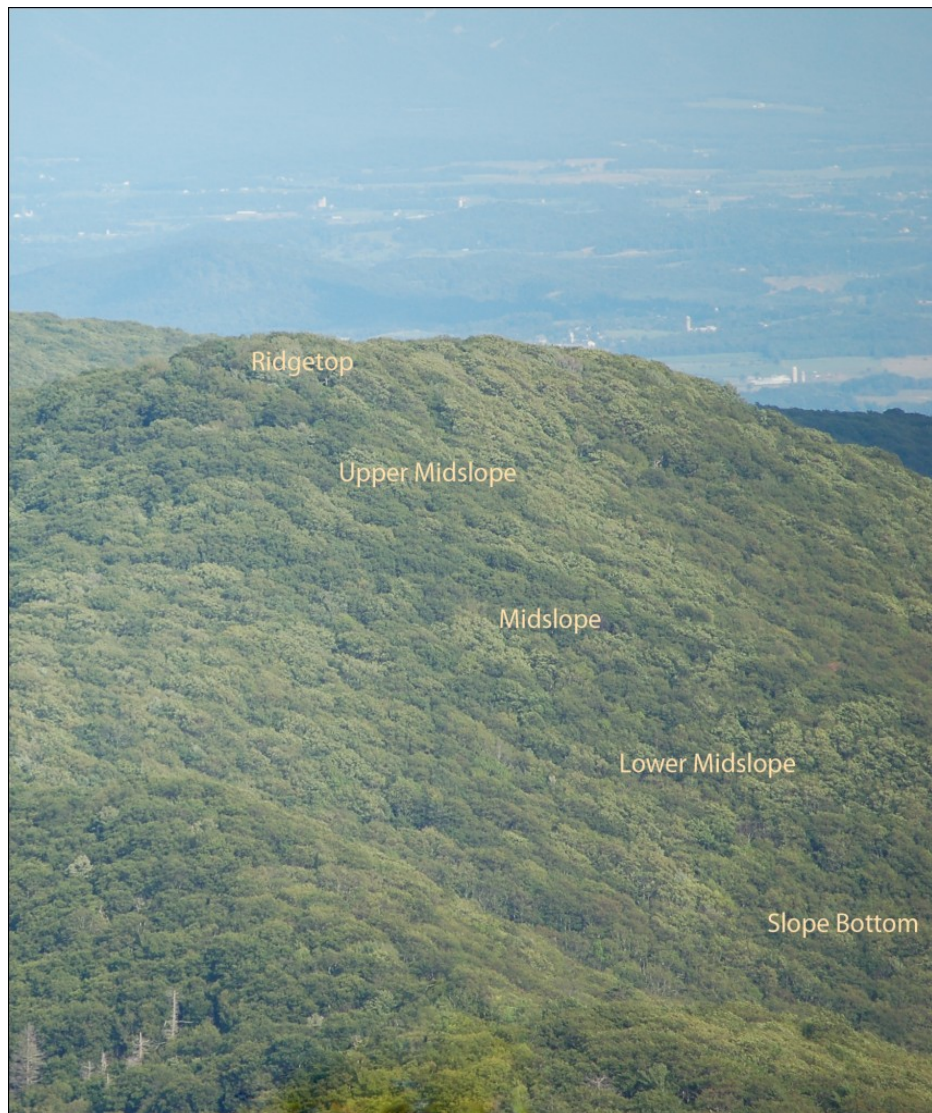


Figure 11: Location of plots within a transect, taken by A. Pipkin, 2009.

Within each plot every ericaceous shrub was cut and a cross section was collected (Figure 12). Shrubs were cut using loppers or a hand saw, depending on the size of the individual sampled. Cross-sections that came from a group of clonal shrubs were aggregated together at the three Virginia sites. Clonal groups were determined by following the root system of the shrubs down to the base. The nature of ericaceous

shrubs and their deep, connected roots makes it difficult to know for certain if all the individuals in a clonal group were collected. It may have been that shrubs divided into separate groups should have been considered a single group. This information was collected to differentiate between clonal shrubs, when possible, so that data on vegetative regeneration could be assessed.



Figure 12: Sampling methods. Removal of shrub cross section with loppers at Goldmine Trail, photo by A. Pipkin, 2009.



In addition to the shrub cross-sections collected from each plot along the transect, the rest of the slope was visually surveyed, and the five largest ericaceous shrub cross-sections were cut from each slope outside of the transect. We extracted these cross-sections to obtain an estimate of the oldest extant of *Kalmia latifolia* at each site.

### Laboratory Methods

After the cross sections were brought to the lab, they were sanded using a belt sander with the smoothest belt being 400 fine grit, and then some were hand sanded using a 3M Microfinishing Film-1200 Grit 9 to 30 Microns, so that the cellular structure of xylem rings were more clearly visible. Each of the individuals was scanned using a Hewlett Packard scanjet 8200 at 2000 dpi resolution. These images were then edited using Adobe Photoshop and imported into WINDENDRO<sup>®</sup>. In WINDENDRO<sup>®</sup> each cross section was evaluated manually for establishment dates by counting the number of rings that were visible on the radii (Figure 13). Cross-dating was not possible due to the inconsistency of ring widths because of the growth form of *Kalmia latifolia*.

### Data Analysis

Individuals were counted at each site according to slope position to measure their abundance. This value was used to estimate density of stems, expressed as stems per hectare. Diameter measurements of all the cross sections were recorded, using calipers. Diameter was used to define the basal area of the shrub layer expressed as square meters per hectare. The basal area at each site and slope position were compared to find if there were significant differences between the groups using the Wilcoxon test that shows differences in the mean between non-parametric data (Ott and Longnecker 2010). The



Establishment was obtained by creating a radius across the cross section and counting the rings within the radius (Figure 13). For most samples only one radius was used to obtain an establishment date, in some cases with larger individual cross sections two or three radii were taken to set a more reliable establishment date. All cross-sections were included in this analysis regardless of pith rot or other factors that made establishment hard to confirm.

When creating age structures, shrubs were grouped by decade because of the uncertainty in ageing individuals. This is a common technique when evaluating dendrochronological data. Histograms show Ericaceae in 10 year bins that represent their decade of establishment. The bins were grouped so that the years 1951-1960 represented the 1950s. The last decade was not composed of a full ten year period due to the time of sampling, so that the 2000s are represented by 2001-2008.

When evaluating age structure data mortality has to be accounted for, assuming that likelihood of mortality increases with time. Age structure data is usually negatively skewed when there is not cessation of establishment. An exponential curve was fit to the data to account for skewness so that attrition of ericaceous shrubs through time could be accounted for. The difference between the exponential curve fit to the age structure and the observed age structure data represents the residuals of establishment (Daniels and Veblen 2004). Residuals estimate decades that have higher or lower than expected establishment. High values are positive while low values are negative. Residuals were calculated in Microsoft Excel.

Data on vegetative regeneration were collected by ageing the oldest individual at each site that belonged to a single clonal group and defining all younger individuals in that group as vegetative regeneration. Then I could calculate the percentage of Ericaceae that were clonal shrubs. It is likely that the estimates representing the percentage of vegetative regeneration are conservative.

The effect of recent fires on shrub establishment was evaluated by comparing residuals of establishment to fire frequency over the past century. The mean number of fire scars per recording tree per decade at each site portrays the frequency and extent of fires. Fire histories were taken from previous dendroecological studies. Correlation between fire and residuals of establishment were evaluated in SPSS. Although fire is the main focus of this study other potentially influential factors that may affect shrub establishment in the southern Appalachian Mountains such as climate and SPB were also evaluated. The Palmer Drought Severity Index (PDSI) was chosen as the most appropriate climate proxy for comparison with age structure residuals. PDSI is a climate proxy that measures how long a drought lasts and the intensity of drought. PDSI is determined through a combination of precipitation and temperature and several other variables such as evapotranspiration. This proxy is cumulative so that the current month's value is dependent upon the weather the month before. Positive PDSI represents wet conditions and negative PDSI represents dry conditions ([NCDC 2010](#)). Assuming that periods with high moisture are favorable for shrub establishment we evaluated residuals of shrub establishment against PDSI. Yearly PDSI and July PDSI were averaged over the entire decade and Spearman's Rho correlations, that tests the dependence of two

variables with non-parametric data, where performed with this proxy and age structure residuals, using SPSS.

The third variable considered was Southern Pine Beetle outbreaks. The forest health atlas provided by the United States Forest Service gives the years in which outbreaks occurred in counties throughout the southeastern states. This record begins in 1960 and ends in 2001 for Virginia and 2000 for Tennessee. Since data is specific to a county the impact of SPB on the particular study sites is not clear. Recent observations of SPB damage were clearly evident at Goldmine Trail. Although the SPB data had very low spatial resolution it was the best available information about this disturbance type. In SPSS SPB outbreaks that occurred at the county level were correlated with residuals of establishment, using a Spearman's Rho correlation.

Duff depth was taken in the field at each plot for the three Virginia sites (Table 3). The correlation between duff depth and basal area along with duff depth and density were calculated using JMP. Duff depth was not take at Goldmine trail, therefore, could not be included in this analysis.

Table 3: Duff measurements. The duff depth (cm) was taken at each transect for the three Virginia Sites

Plot	Griffith Transect 1 (cm)	Knob Transect 2 (cm)	Brush Transect 1 (cm)	Mountain Transect 2 (cm)	Reddish Transect 1 (cm)	Knob Transect 2 (cm)	Average
Ridgetop	1	4	7	3	7.5	7	4.9
upper midslope	5	7	6	1.5	5.5	10	5.8
midslope lower	7	3	6.5	1.5	7.5	7	5.4
midslope slope	5	4	4	2	6	6	4.5
bottom	1	1	2.5	1	1	2	1.4
Average	3.8	3.8	5.2	1.8	5.5	6.4	4.4

## CHAPTER IV

### RESULTS

#### Composition of the Ericaceous Community

There were three species of ericaceous shrubs collected at four different study sites. *Kalmia latifolia* was the dominant species on all sites. *Rhododendron maximum* was seen at all sites but only fell within the boundaries of the plots at Goldmine Trail and Griffith Knob. *Pieris floribunda* was only collected at Reddish Knob. There were 829 cross-sections amassed from the four sites. There were 258 individuals collected at Goldmine Trail, 145 at Griffith Knob, 149 at Brush Mountain and 277 at Reddish Knob.

#### Density and Basal Area

Density and basal area varied across the different sites and slope positions. Reddish Knob has the highest density of ericaceous shrubs and Griffith Knob has the lowest density (Figure 14). Goldmine Trail had the highest basal area and Griffith Knob had the lowest basal area (Figure 15). If *Rhododendron maximum* is excluded than Reddish Knob has the highest basal area. The results of a Wilcoxon statistical test shows there are no significant differences (Table 4) in the density of ericaceous shrubs found between the sites. There is, however, a significant difference between the basal area of ericaceous shrubs found between sites ( $p=0.0052^*$ ). There are no significant differences in density or basal area of *Kalmia latifolia* between the sites. Because of the large stature of *Rhododendron maximum* the basal area is higher at Goldmine Trail than Reddish Knob although the former had a lower density.

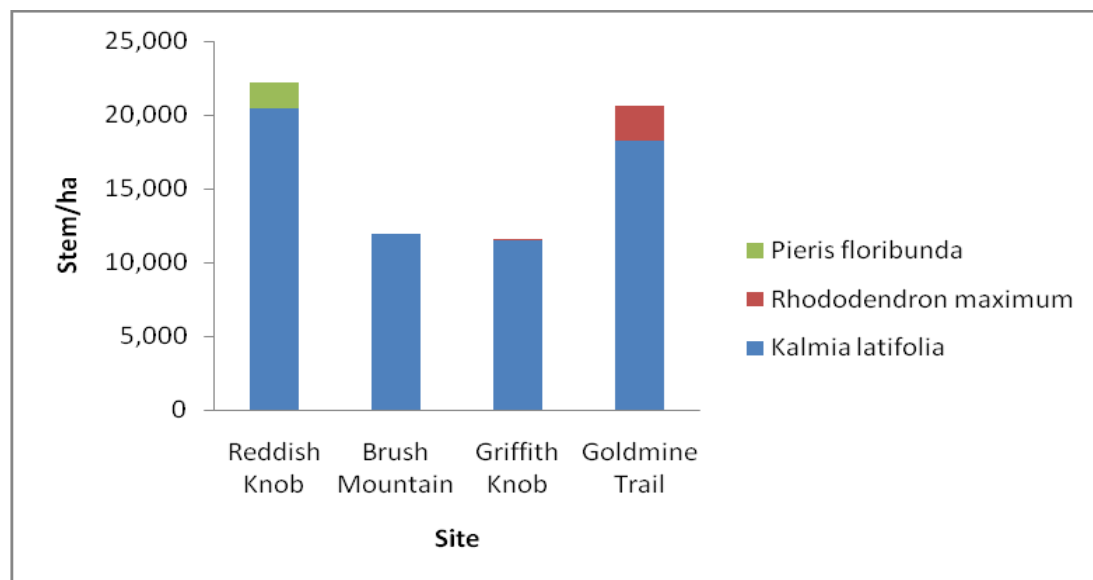


Figure 14: Density of ericaceous shrubs at each site. The density and composition of ericaceous shrub species at each site reported in ericaceous stems per hectare.

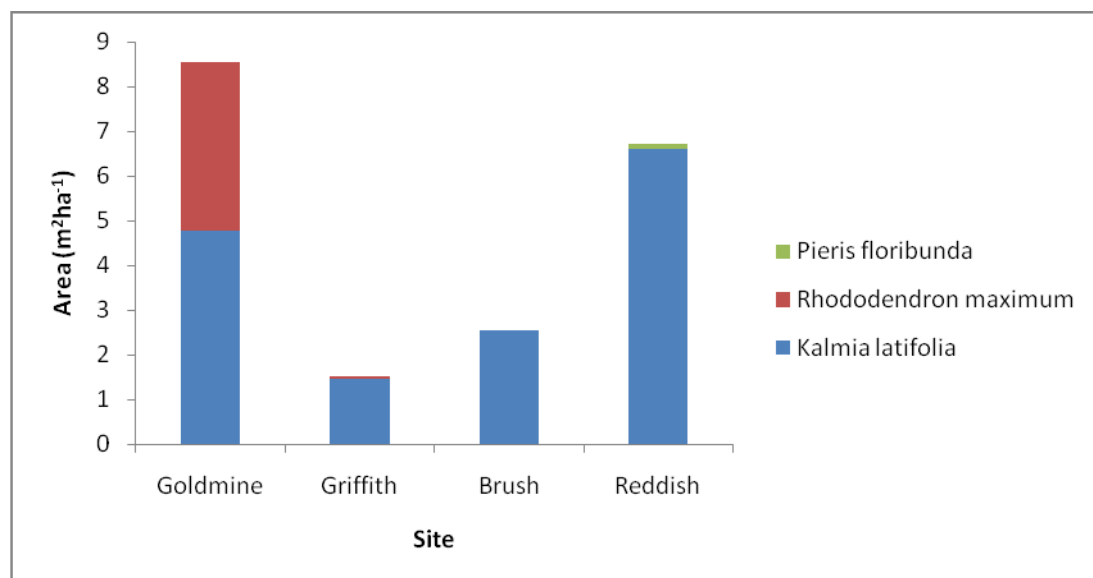


Figure 15. Basal area of ericaceous shrubs at each site. The basal area and composition of ericaceous shrub species at each site reported in  $\text{m}^2\text{ha}^{-1}$ .



Table 4: Results of the non-parametric Wilcoxon statistical test comparing ericaceous shrubs between sites.

	Density		Basal Area	
	Ericaceous shrubs	<i>Kalmia latifolia</i>	Ericaceous shrubs	<i>Kalmia latifolia</i>
p-value	0.1716	0.4481	0.0052*	0.0577
Chi-square (test statistic)	5.0025	2.6538	12.743	7.4957
Degrees of Freedom	3	3	3	3

The density of Ericaceae at each slope position reveals a peak density at the mid-slope (Figure 16). This peak remains constant when examining the basal area at each slope position. Because of *Rhododendron maximum*, the slope bottom has a relatively high basal area (Figure 17). The basal area and density of *Kalmia latifolia* are highest in the middle and upper slope positions. *Pieris floribunda* has the highest density and basal area on ridgetop plots. There is no significant difference in basal area of ericaceous shrubs between the five slope positions (Table 5). There are significant differences in the basal area and the density of *Kalmia latifolia* between the four sites. There is a significant difference ( $p=0.0371^*$ ) between the density of ericaceous shrubs collected on the five slope positions.

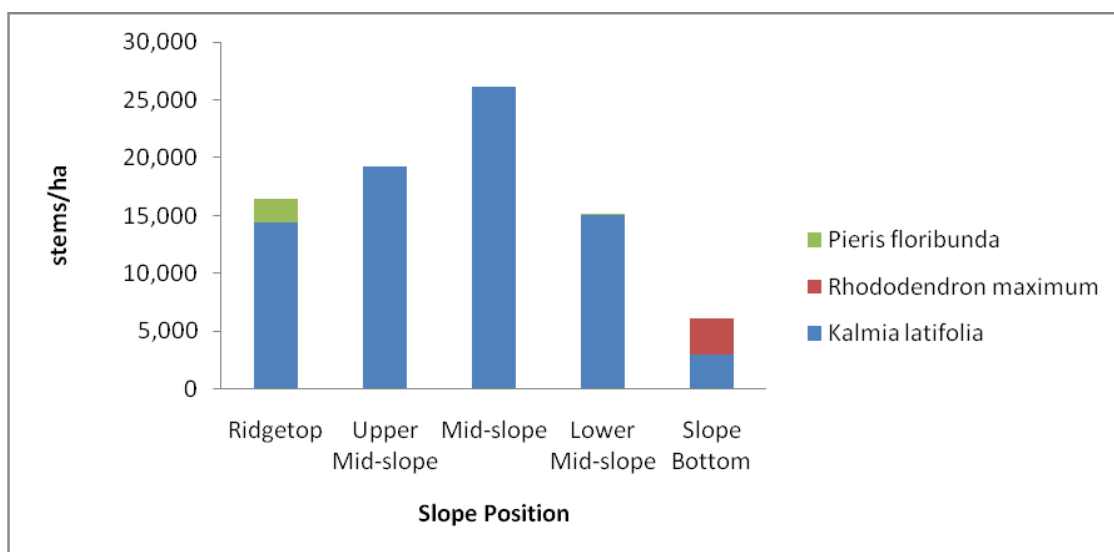


Figure 16: Density of ericaceous shrubs at each topographic position. The density and composition of ericaceous shrub species at each slope position for all sites reported in ericaceous stems per hectare.

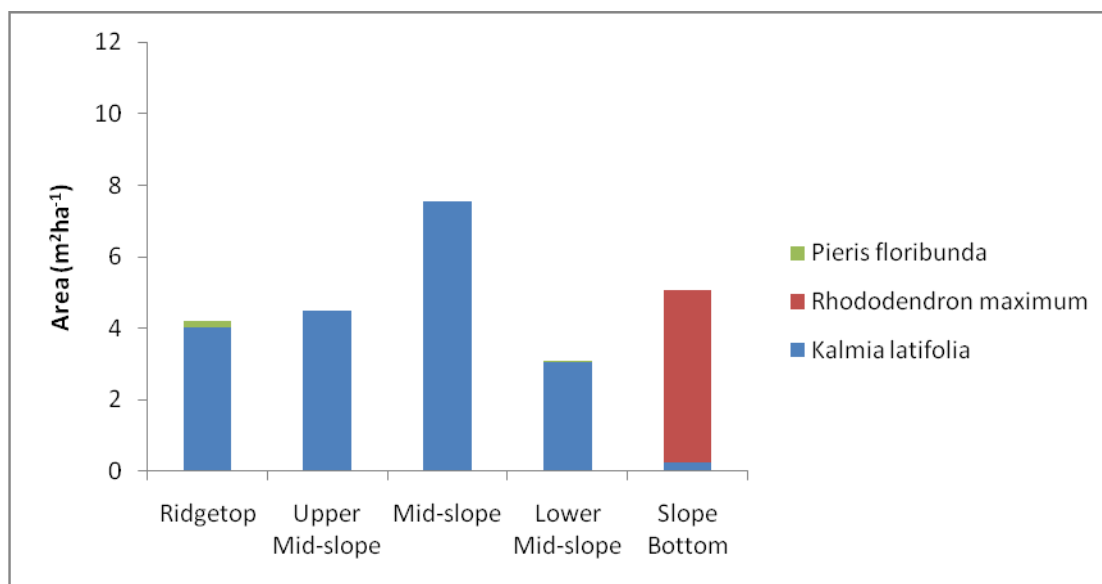


Figure 17: Basal area of ericaceous shrubs at each topographic position. Basal area of ericaceous shrubs collected on each site by slope position.

Table 5: Results of the non-parametric Wilcoxon statistical test comparing ericaceous shrubs between topographic positions.

	Density		Basal Area	
	Ericaceous shrubs	<i>Kalmia latifolia</i>	Ericaceous shrubs	<i>Kalmia latifolia</i>
p-value	0.0371*	0.0067*	0.1714	0.0014*
Chi-square (test statistic)	10.2036	14.1951	6.3976	17.7038
Degrees of Freedom	4	4	4	4

### Duff

There was a significant ( $p=0.001^*$ ) correlation of 0.53001 between the density on each plot and the thickness of the duff layer. In addition to the density there was a significant ( $p=0.001^*$ ) correlation of 0.61352 between the basal area on each plot and the thickness of the duff layer.

### Age Structure

Rudimentary statistics about *Kalmia latifolia* show that the oldest median population is found at Reddish Knob (Table 6). Goldmine trail has the greatest age range. At Reddish Knob and Brush Mountain there has been no establishment in the last three years. The ages of the largest cross-sections show that *Kalmia latifolia* can live to be around 100 years old in the study area (Figure 18).

Table 6: Descriptive statistics for *Kalmia latifolia*. The median, oldest and youngest cross sections collected at each site.

Sites	Oldest	Youngest	Median
Goldmine Trail	1927	2008	1994
Griffith Knob	1951	2008	1994
Brush Mountain	1946	2005	1987
Reddish Knob	1929	2005	1982

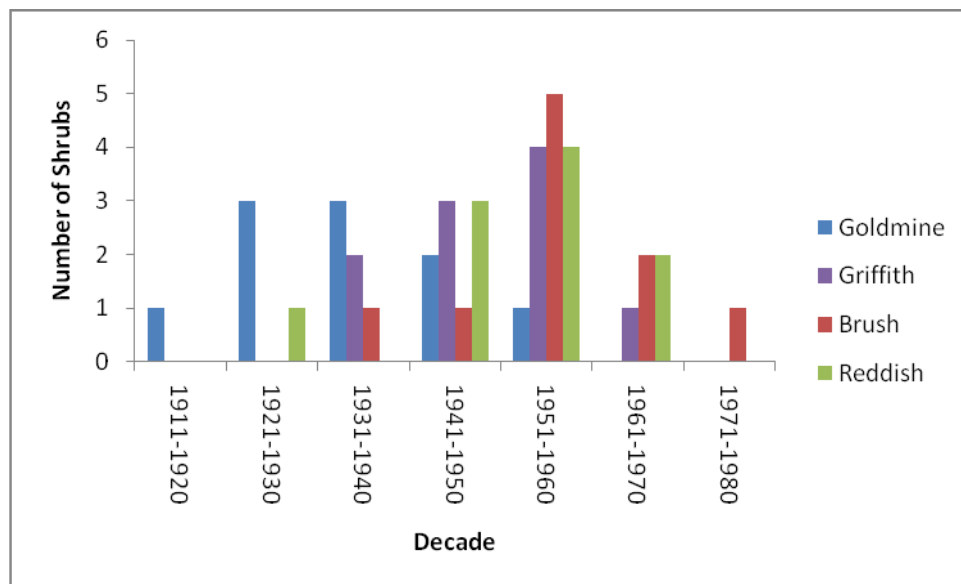


Figure 18: Age structure of the largest cross-sections. The oldest individuals collected from each site offer a representation of the oldest extant of the species in the study area.

Age structures for each site represent the establishment by decade of ericaceous shrubs from 1921 until 2008 (Figure 19). The residuals of establishment (Figure 20) represent years with higher than average establishment and lower than average

establishment. All cross-sections collected within the plots over the study area are included in these age structure graphs. The graphs have an exponential trendline fitted to the data.

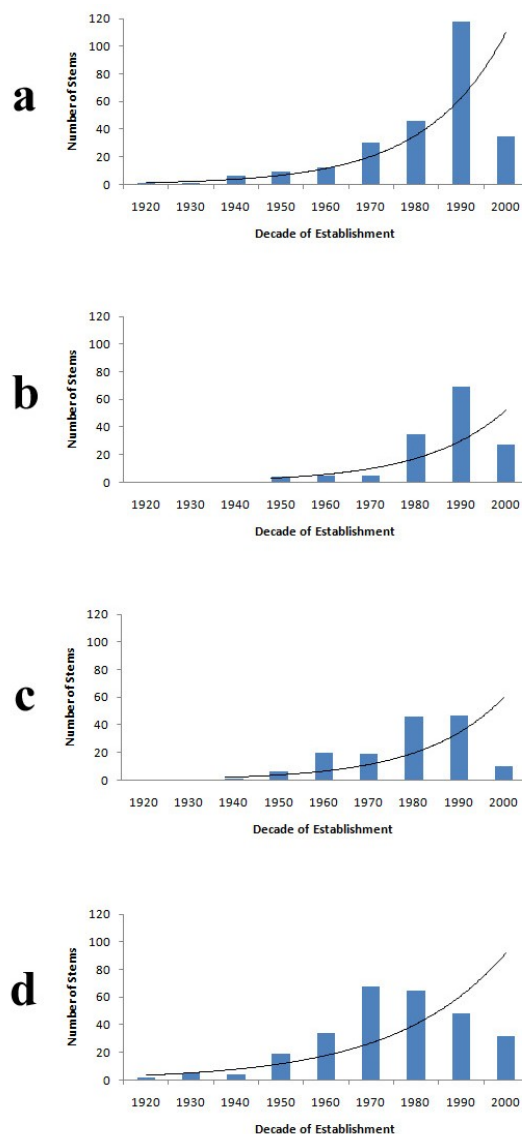


Figure 19: Age structures for ericaceous shrubs. a) Goldmine Trail b) Griffith Knob c) Brush Mountain d) Reddish Knob. Age structures have an exponential curve fit to the data.

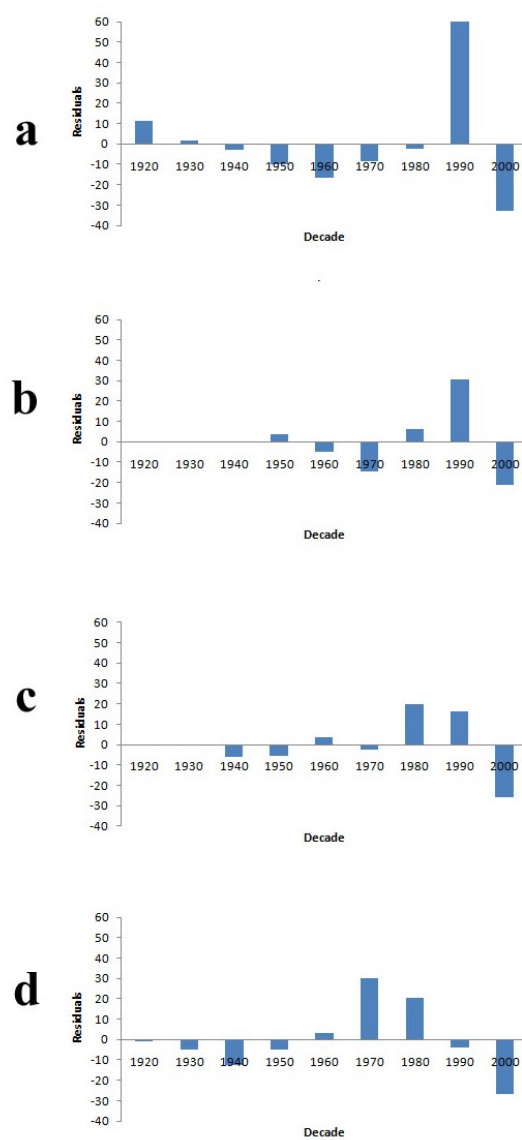


Figure 20: Residuals of ericaceous shrubs. Residuals were determined from the exponential curve a) Goldmine Trail b) Griffith Knob c) Brush Mountain d) Reddish Knob.

Establishment of cross-sections is classified by site and slope position for Goldmine Trail (Figure 21), Griffith Knob (Figure 22), Brush Mountain (Figure 23) and Reddish Knob (Figure 24). At Goldmine Trail there were only *Rhododendron maximum* found at the slope bottom. These *Rhododendron maximum* spanned a large range of ages and were growing in a well developed shrub thicket. The highest concentration of ericaceous shrubs is found at the three higher slope positions, which consisted of only *Kalmia latifolia*. Griffith Knob, has the lowest overall shrub cover. Griffith Knob has no ericaceous shrubs in the ridge-top and a single *Rhododendron maximum* at the slope bottom. At this site the oldest ericaceous shrubs are in the lower mid-slope and mid-slope. Brush Mountain has no individuals at the slope bottom and a high concentration in the mid-slope. The oldest cross-sections were collected at the mid-slope. Brush Mountain is comparatively peculiar because there are few Ericaceae found at the upper mid-slope while larger older Ericaceae are found on the surrounding slope positions. Reddish Knob is the only site where *Pieris floribunda* was collected and *Kalmia latifolia* was found at every slope position. Like the other three sites Reddish Knob has the oldest individuals found at the mid-slope and younger individuals away from the center of the slope. There were few young cross-sections collected at this mid-slope position compared to cross-sections over 30 years of age.

Across all sites the oldest cross-sections were collected in the mid-slope. The youngest *Kalmia latifolia* are found away from this slope position. This coincides with the basal area for these slope positions. The highest basal area has the oldest cross-sections and the lowest basal area has the youngest cross-sections.

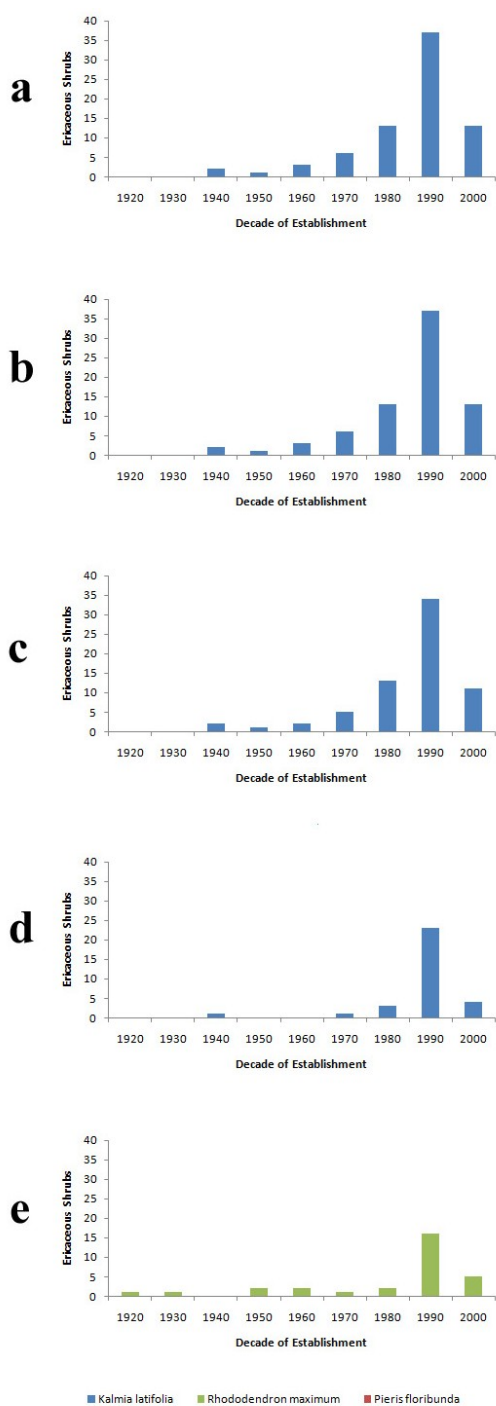


Figure 21: Age structure of Goldmine Trail by topographic position. a) ridgetop b) upper mid-slope c) mid-slope d) lower mid-slope e) slope bottom.



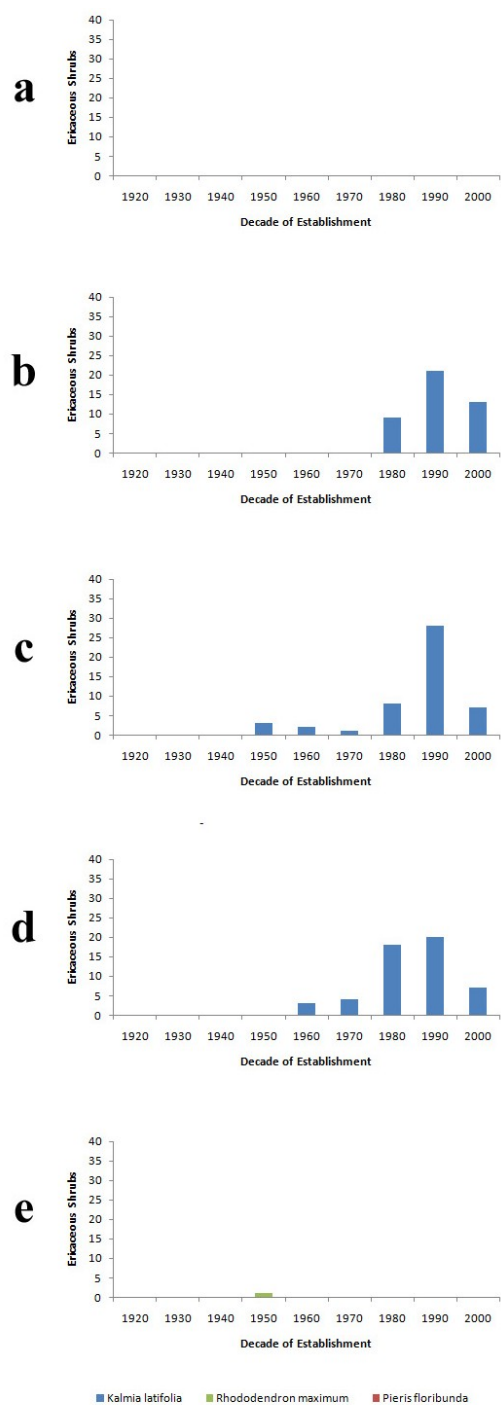


Figure 22: Age structure of Griffith Knob by topographic position. a) ridgetop b) upper mid-slope c) mid-slope d) lower mid-slope e) slope bottom.

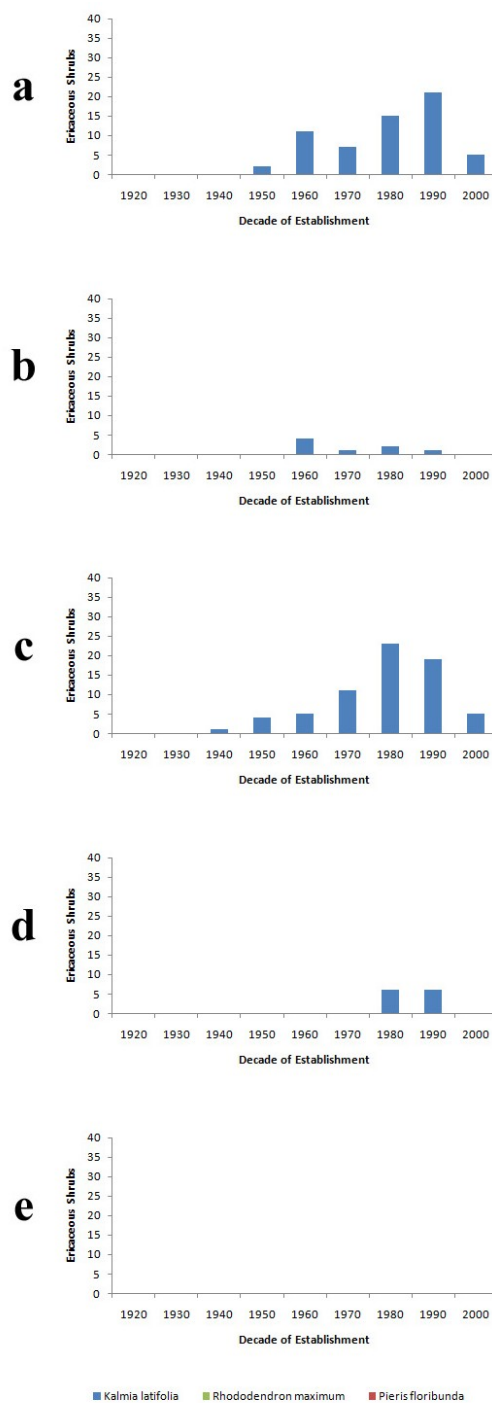


Figure 23: Age structure of Brush Mountain by topographic position. a) ridgetop b) upper mid-slope c) mid-slope d) lower mid-slope e) slope bottom.

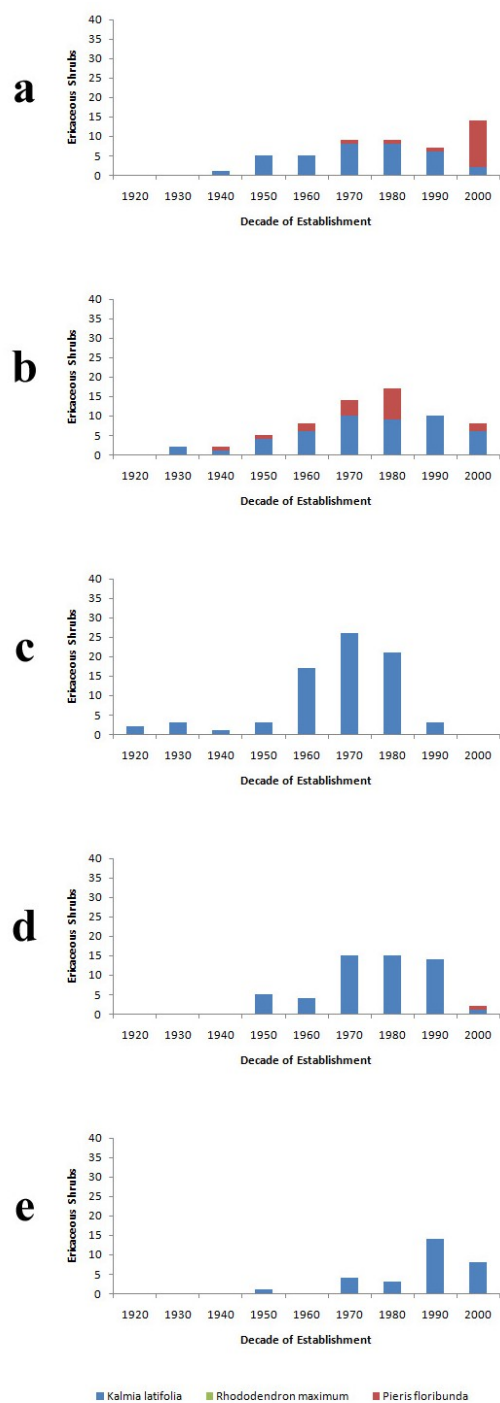


Figure 24: Age structure of Reddish Knob by topographic position. a) ridgetop b) upper mid-slope c) mid-slope d) lower mid-slope e) slope bottom.

### Vegetative Regeneration

An age structure was produced for the oldest cross-sections in a group of clonal shrubs (Figure 25). Goldmine trail is excluded in this analysis here because clonal shrubs were not grouped together at this site. This analysis was performed to observe the age structure without the shrubs that were clearly vegetative offspring. The residuals (Figure 26) show the last decade is still below average and the 1990s are still above average, however, these differences are more subdued than the data including vegetative regeneration (Figure 20). Brush Mountain has had no new recruitment and the establishment of shrubs in the last decade has been the result of vegetative reproduction, which hasn't occurred since 2005 (Table 6).

Ericaceous shrub age structures of vegetative regeneration (Figure 27) show that the majority of these individuals are relatively young. Griffith Knob's age structure has the youngest group of vegetative regeneration beginning in 1980. Brush Mountain almost has a constant increase yet there is reduced sprouting in the most recent decade. Reddish Knob seems to have a persistent pattern of sprouting for the past 40 years with a few older sprouts.

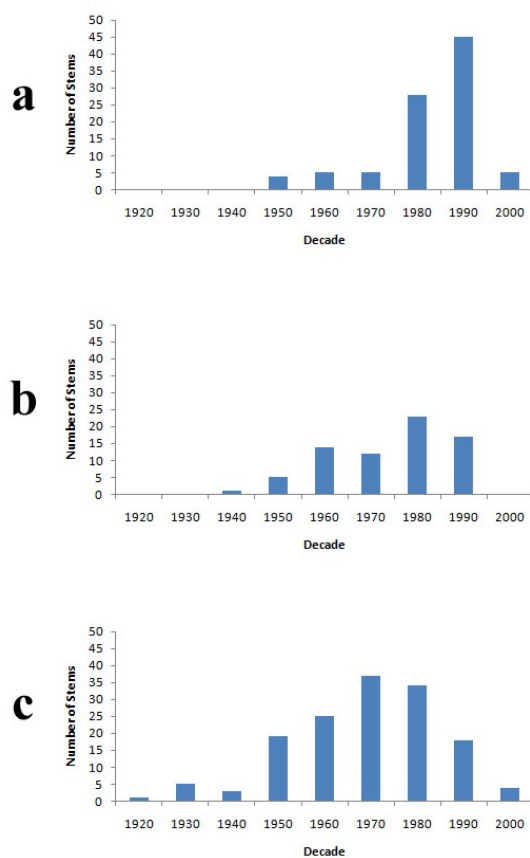


Figure 25: Age structure of oldest cross-sections in a clonal group. The oldest cross-sections in a clonal group of ericaceous shrubs represent the beginning of establishment for that group of individuals. a) Griffith Knob b) Brush Mountain c) Reddish Knob.

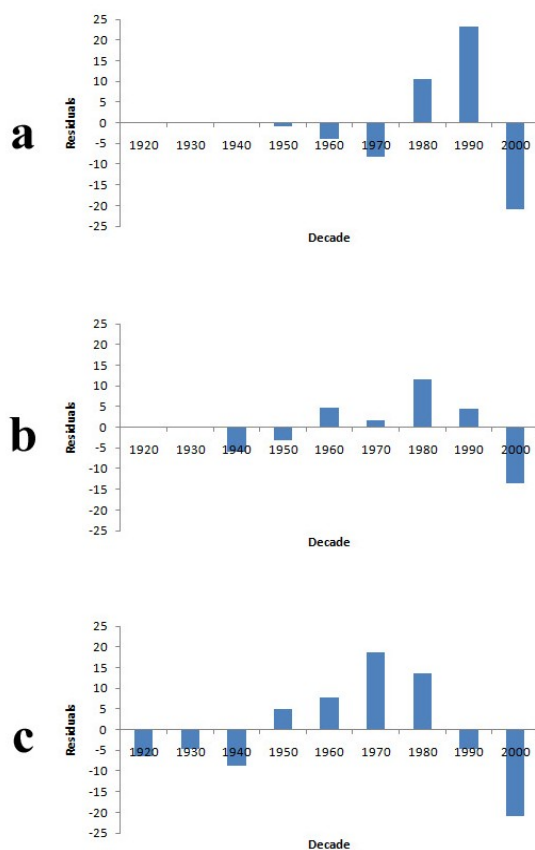


Figure 26: Residuals of the oldest cross-sections in a clonal group. Residuals were taken from an exponential trendline fitted to the age structure data in a clonal group of ericaceous shrubs. a) Griffith Knob b) Brush Mountain c) Reddish Knob.

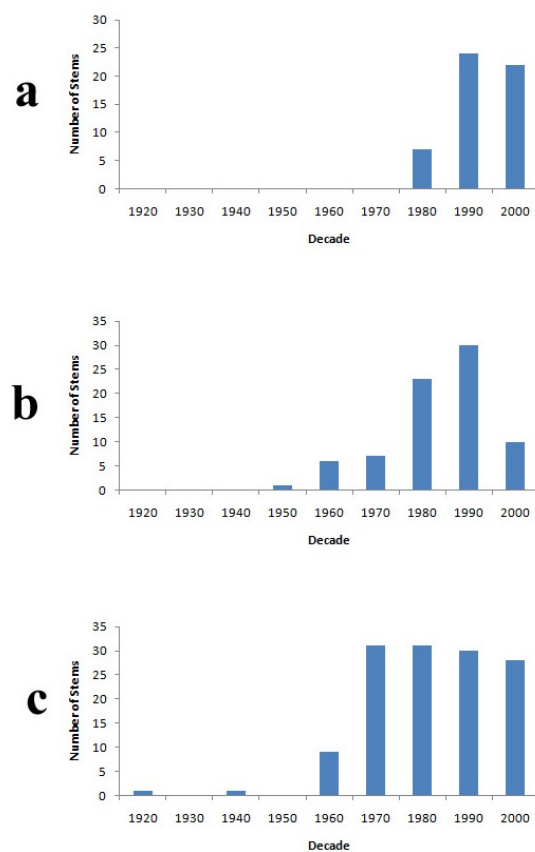


Figure 27: Vegetative regeneration of ericaceous shrubs. a) Griffith Knob b) Brush Mountain c) Reddish Knob.

The percentage of vegetative reproduction that occurs on each site shows that Griffith Knob (37%) has had the lowest amount of vegetative reproduction (Figure 28). Reddish Knob (47%) and Brush Mountain (52%) have nearly an equivalent amount of vegetative reproduction. *Kalmia latifolia* is sprouting and over time and without disturbance is regenerating vegetatively.

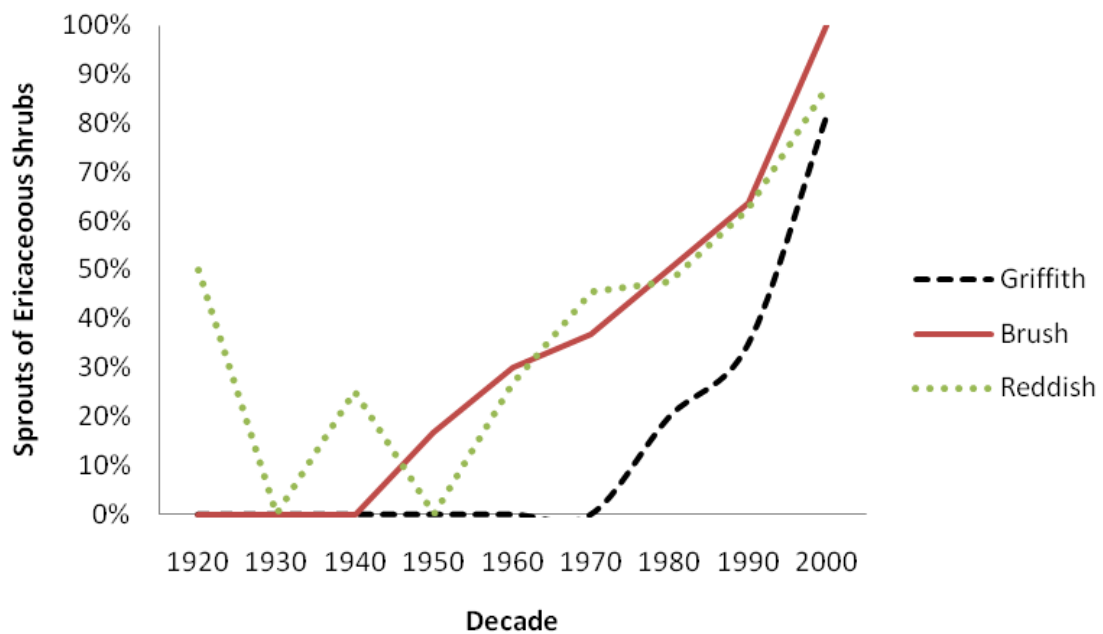


Figure 28: Percentage of shrubs that were reproduced vegetatively.



### Fire and Ericaceous Shrub Establishment

Establishment residuals over the past century were compared to the mean number of fire scars per recording tree per decade at each site. The establishment graphs show that in some cases fires, either because they did not burn the entire area or they were not intense enough, did not remove all the Ericaceae from the landscape (Figure 29). There does seem to be a relationship between fire events and shrub cover. The only site with a significant correlation was Reddish Knob that had a high Pearson correlation value of 0.832 (Table 7). However, since there were only four data points the data set may be too sparse for this type of analysis.

### Climate Interactions

Residuals of establishment for the past 90 years were compared with PDSI . When examining the relationship between the residuals of establishment and the decadal PDSI as well as decadal July PDSI there was no significant relationships between any of the sites. Although Spearman's Rho had no significant correlations there does seem to be a negative trend with July PDSI (Table 8).

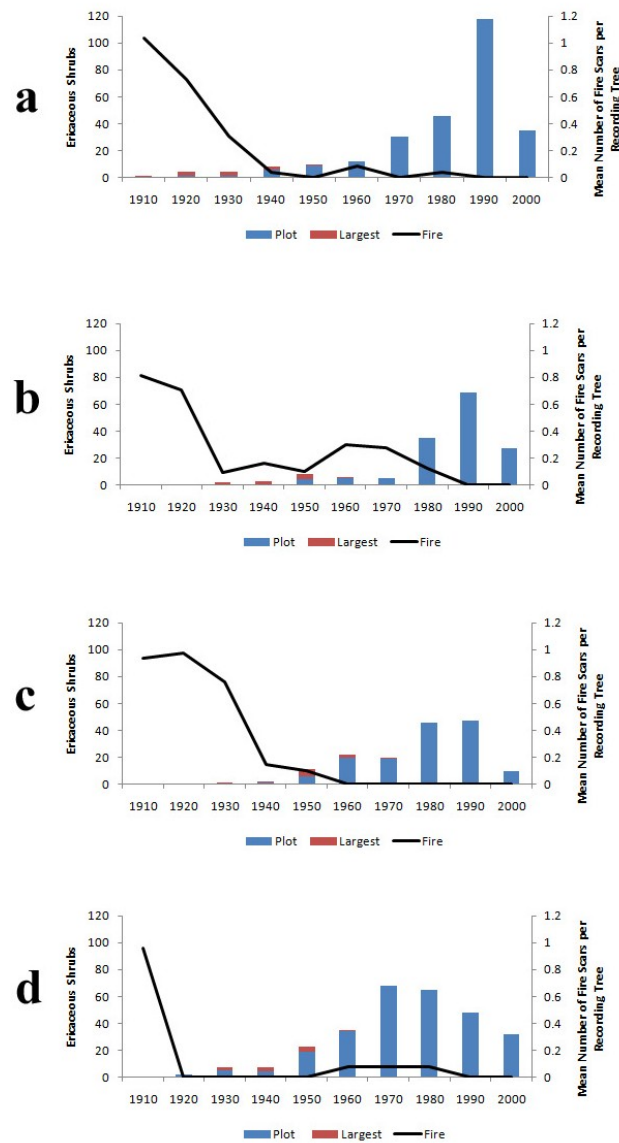


Figure 29: Ericaceous shrub density compared with fire frequency. Fire frequency (mean number of fire scars for each recording tree for each decade) and shrub establishment over the past century that includes the 10 largest cross-sections collected at each site and the plot data.

Table 7: Correlation between fire and age structure residuals. The Pearson Correlation and p-value for the relationship between fire events and residuals at each site.

Site	Pearson Correlation	p-value
Goldmine Trail	0.56	0.895
Griffith Knob	-0.453	0.259
Brush Mountain	-0.294	0.48
Reddish Knob	0.832	.010*

Table 8: Correlation of PDSI and age structure residuals. The Spearman's rho correlation values of PDSI and ericaceous age structures.

Site	July PDSI Pearson Correlation	July PDSI p-value	Yearly PDSI Pearson Correlation	Yearly PDSI p- value
Goldmine Trail	-0.083	0.831	0.067	0.865
Griffith Knob	-0.288	0.452	-0.153	0.695
Brush Mountain	-0.636	0.544	0.066	0.130
Reddish Knob	0.192	0.62	-0.159	0.683

### Southern Pine Beetle Significance

The record for southern pine beetle outbreaks only goes back to the 1960s and stops in 2000 for Goldmine Trail and 2001 for the Virginia sites. There is no significant difference between decades with or without SPB outbreaks for each county and residuals of establishment. Evaluating only Goldmine Trail, which has the highest number of SPB outbreaks, the Spearman's rho correlation value is -0.316 and the p-value is 0.684. Although this relationship is not significant, observational evidence suggests that this disturbance type may be a contributing factor and warrants further investigation.

## CHAPTER V

### DISCUSSION

#### Ericaceous Shrub Characteristics across Study Sites

*Kalmia latifolia* is the most abundant shrub found in Appalachian Mountain xeric pine-oak forests (Whittaker 1956), and was the most abundant ericaceous shrub at all of the sites in this study. Ericaceous shrub density differed significantly across slope positions. Basal area of ericaceous shrubs also differed significantly between sites. This research aims to explore possible reasons for the differences in ericaceous shrub density and basal area at different sites and slope positions within the central and southern Appalachian xeric pine-oak forests.

The density of stems found on this site were within the normal ranges throughout the Appalachian Mountains. On the four sites there was a range of density from 11,520 stems/ha to 20,480 stems/ha. Other research has found higher and lower densities in undisturbed xeric pine-oak forests (Clinton et al. 1993; Ducey et al. 1996; Elliott et al. 1999). Elliott et al. (1999) found 8,810 stem/ha (*K. latifolia* and *R. maximum*). Clinton et al. (1993) found 21,525 stems/hectare (*K. latifolia*). Ducey et al. (2006) found densities of 22,500 stems/ha and 35,700 stems/ha (*K. latifolia*). These studies measured stems per hectare but did not take into consideration the basal area. Ducey et al. (1996) found that shrub densities increased after fire but much more with a low intensity fire. So although there may have been an increase in stems the overall basal area was likely reduced.

Across all sites the highest density and the highest basal areas were found at the mid-slope position. The lower mid-slope positions had a lower density of ericaceous shrubs, which were younger than those found at higher slope positions. The slope bottom had the lowest shrub density. Due to the large stature of *Rhododendron maximum* found only at the slope bottom this topographic position had a relatively high basal area. There was a decreasing gradient from south to north of *Rhododendron maximum*. Where *Kalmia latifolia* and *Rhododendron maximum* overlap the latter is often found on the lower slopes closer to streams and *Kalmia latifolia* is often found further away from water sources (Day and Monk 1974). The differences between slope position is most likely the result of a competitive advantage of *Rhododendron maximum* over *Kalmia latifolia* in mesic sites with more shade, rather than the optimal conditions for *Kalmia latifolia* (Day and Monk 1974). This is evident on Reddish Knob where no *Rhododendron maximum* were collected and *Kalmia latifolia* occupied the slope bottom.

Goldmine Trail had the highest basal area, which was in part, the result of the large *Rhododendron maximum* collected on the site. During the last century Goldmine Trail's fire history is slightly more active than Brush Mountain and less active than Reddish Knob. The last fire at Goldmine Trail occurred in 1982 but this was recorded in a stand adjacent to the transects in this study and may not have affected these transects. It appears from the establishment patterns that the last fire to affect these transects was the fire that occurred in 1960. At Goldmine Trail the fewest individuals were found at the lower mid-slope, where the transition from *Kalmia latifolia* to *Rhododendron*

*maximum* occurs. Goldmine trail had a gentler slope than the other three sites. The topography at Goldmine trail may account for the similar density and age structure found at the three highest slope positions. The low abundance of shrubs at the lower mid-slope seemed to be the result of the gap between the transition zones of *Rhododendron maximum* and *Kalmia latifolia* in which few Ericaceae occurred. Although I am sure ericaceous shrubs can establish at this slope position I speculate that this gap occurred because it was too dry for *Rhododendron maximum* and too shaded under the dense forest canopy for *Kalmia latifolia*. However, if *Tsuga canadensis* or *Pinus spp.* are found at this slope position and suffer mortality due to biological disturbances then shrubs are likely to increase.

Griffith Knob had the lowest stems per hectare and the lowest total basal area of any site. This site also had the most active fire regime after fire suppression began. The density of stems collected was similar to what was collected at Brush Mountain yet the basal area is much lower. The most recent fire (1985) likely reduced the basal area of Ericaceae on this site. The small size of these cross-sections is most likely a result of this fire, this assertion is supported by their young age. The highest concentration and oldest shrubs at Griffith Knob were found in the middle slope positions, with no individuals found at the ridge-top, and only a single *Rhododendron maximum* found at the slope bottom. Moving away from the mid-slope shrubs were younger and less dense, suggesting expansion of *Kalmia latifolia* into the upper and lower slope positions since fire suppression.

At Griffith Knob the age structure shows that the majority of shrubs between the 1980s and 2000s were vegetatively reproduced (Figure 25) and over 90% of the cross-sections that were collected established at this time. Vegetative reproduction had perhaps been occurring on this site all along but many of the Ericaceae may have been top-killed after the last fire on Griffith Knob in 1985. This would have caused an increase in sprouting from the underground reserves after this time. All the individuals at the upper mid-slope dated after the 1980s and there were no individuals at the ridge-tops or slope bottoms. Wildfires unlike milder prescribed fires likely burned hotter on the ridge-tops, possibly killing underground parts of the plant and the frequency of fire may have been enough to deter encroachment into the slope bottom.

On Brush Mountain there was a relatively low number of ericaceous shrubs, similar to Griffith Knob, however the basal area was much higher. This is possibly the result of fire occurring earlier on Brush Mountain. It seems the 1956 fire that occurred on Brush Mountain may have influenced shrub cover. There is only one shrub found at the mid-slope position that is older than this fire event. The stems on Brush Mountain tended to be larger and as a result the basal area was higher. Brush Mountain like Griffith Knob had no *Kalmia latifolia* at the slope bottom, but unlike the other three sites the oldest thickets were at the ridgetop and mid-slope. On Brush Mountain there seemed to be two main thickets at the ridgetop and mid-slope with a gap in between that was beginning to see the early signs of encroachment. The youngest Ericaceae were found further down slope.



Reddish Knob had the highest density and basal area of *Kalmia latifolia*. *Kalmia latifolia* on this site were larger than the other three sites and much more contiguous creating a “mid-story forest” that was not an obvious feature at the other three sites. Anderson (1988) reported a similar phenomenon with tall old *Kalmia latifolia* having continuous cover and low regeneration underneath. *Kalmia latifolia* was found at every slope position. At Reddish Knob, like the other three sites, the oldest individuals were around the mid-slope. Reddish Knob had a drop in vegetative reproduction around the 1950s that seems to correspond to a fire event during the following decade. Although Reddish Knob has had recent fires it seems as though they were small and isolated or possibly mild in intensity. The fires only scarred a single pine cross-section that was collected (Aldrich 2011). Mild surface fires may be effective in reducing leaf litter but not removing ericaceae. The last widespread fire that occurred at this site was in 1913, this is earlier than the last widespread fire on the other sites. After this time there were no signs of fire besides on a single individual tree that recorded four fires. Because these fires were only recorded by one tree, they may have had limited spatial extent (Grissino-Mayer et al. 2004), or very mild severity. It seems likely that this later fire event did not influence the basal area or abundance of Ericaceae on Reddish Knob. If the fire did reach these stands it may have not been hot enough to remove shrubs from the landscape and possibly only killed the youngest individuals which may account for the low number of young individuals at this site. The age structure on Reddish Knob is representative of an age structure without influential fires for nearly a century. The oldest individuals were found at the mid-slope, here most of the Ericaceae are around, 30 to 50 years old,

with comparatively little establishment afterwards (Figure 23). It seems that this is because the Ericaceae have formed a mature shrub layer that prevents further establishment, unless there is a disturbance knocking one down or opening the “ericaceous canopy.”

Although fire suppression in the southern and central Appalachians began rather abruptly each of these sites has a unique interaction and history with fire. After fires occurred on these sites ericaceous shrub density was reduced. Fires influence ericaceous shrub cover. Once a threshold for shrub density is crossed it may be much more difficult to return the forest back to the low shrub cover state by fire alone (Brose and Waldrop 2010).

#### Ericaceous Shrub Characteristics across the Study Area

The age structure of Ericaceae (Figure 21, Figure 22, Figure 23 and Figure 24) can tell us the age of thickets and suggests the time when *Kalmia latifolia* established and possibly if it has encroached into an area where Ericaceae were previously limited. Ericaceous shrubs were on the Appalachian landscape before fire suppression, but coverage has increased possibly because of the absence of fire (Dobbs and Parker 2004). With frequent fires it has generally been assumed that there were fewer ericaceous shrubs on the landscape. The oldest *Kalmia latifolia* collected dated back to approximately 1915 making its age at the time of removal about 94 years. This coincides with establishment dates from other dendrochronological studies in the Appalachian Mountains (Brose and Waldrop 2010; Aldrich 2011).

When evaluating age structure graphs the appearance of the J-shaped distribution that commonly arises may be produced for a couple of different reasons. If populations are stable or if there is enhanced recruitment by younger individuals an age structure will appear J-shaped (if the order on the x-axis starts with the oldest to youngest from left to right). This makes it difficult to interpret if the data shows a population that is increasing recruitment and cover through time or if the population is stable. One thing this does tell us is that the population is not declining. If the goal for xeric Appalachian forests is to increase pine establishment then management strategies should be put into place to reduce ericaceous shrubs. If ericaceous shrubs were declining you would not expect to see a J-shaped curve.

Because a static age structure was created the residuals (Figure 20) account for attrition and provide information about decades with high or low establishment (Daniels et. al, 2004). The residuals suggest above average levels of establishment for the 1990s for Brush Mountain, Griffith Knob and especially Goldmine Trail. In the most recent decade the levels of establishment are much lower than expected at all sites. In part, this may be the result of the eight year span of the last bin versus the ten year bin for all the other decades.

The oldest individuals from the plot data are found at the mid-slope position on all three slope positions. As you move up slope and down slope the cross-sections are younger and density decreases. This suggests that the mid-slope may provide the optimum habitat for *Kalmia latifolia* under a more frequent burning regime (Figure 30).

It could also be possible that under fire suppression that the mid-slope remains the optimum environment and individuals can live longer at this slope position.

There are a number of reasons that could account for the high shrub cover and old ages found at the mid-slope compared to the other slope positions. Research on fire in the Appalachian Mountains suggests that the hottest and most intense fires occur on the ridge-top where the most fire-tolerant species are found (Zobel 1969; Waldrop and Brose 1999). There is evidence that intense fires reduce cover of Ericaceae more than milder surface fires (Ducey et al. 1996).

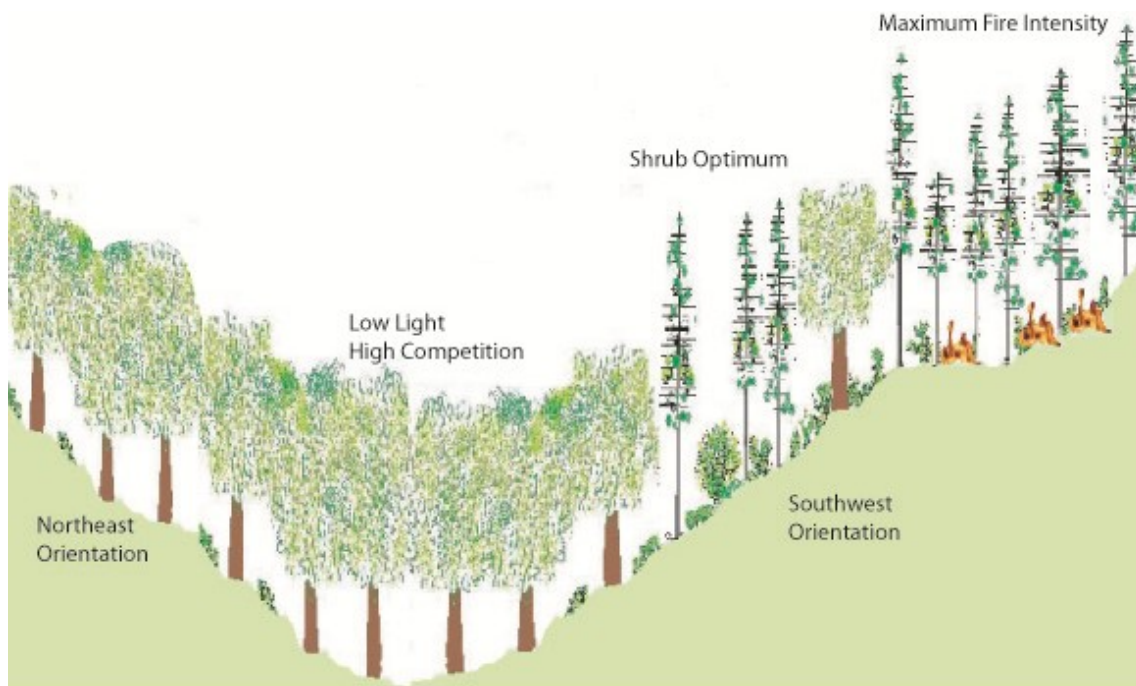


Figure 30: Conceptual model of shrub optimum. Historically density of *Kalmia latifolia* may have been highest at the mid-slope position. This is because at the ridge-top fires are more severe and at the slope bottom there was more competition and less light availability in the higher density stands.

As you move further down the slope into the lower mid-slope there would be more competition with hardwood species after a fire event. The dry open conditions that are found higher on the ridge-top would not be as prevalent. Ericaceous shrubs would have been competing with other hardwood sprouts on the lower slope and on the north and northeast facing slopes, reducing their abundance because of higher competition, due to lower light and lower resources because of the higher arboreal density. With more competition further down slope and a history of more intense fires at the ridgetop it seems reasonable that the densest midstory would have been found at the mid-slope under the historic fire regime. Reduced fire activity would have permitted increases towards the upslope and expansion into the lower slope positions where shrubs were previously not as prevalent.

The only site where *Kalmia latifolia* occurs in the slope bottom is on Reddish Knob. At Reddish Knob at the slope bottom there is a single individual that dates back to the 1950s with most of the other Ericaceae establishing in the 1970s. This suggests that prior to fire suppression, while *Rhododendron maximum* might have been common close to slope bottoms as is evident at Goldmine Trail, *Kalmia latifolia* was often not found at lower slope positions. In the absence of fire *Kalmia latifolia* has extended into areas where it was not previously located. The pattern of *Kalmia latifolia* establishment at these sites suggests an encroachment of ericaceous shrubs from the mid-slope position down into the lower slope positions, in the absence of fire. This pattern is most evident at Reddish Knob, the most fire suppressed site.

The decrease in establishment during the last decade is a prominent feature of the age structure graphs. The last bin of the age structure graphs only represented eight years and this is partially responsible for the low number of Ericaceae represented. The two other studies that collected dendroecological data on *Kalmia latifolia* and the ericaceous shrub layer did not collect the youngest individuals (Brose and Waldrop 2010; Aldrich 2011), therefore there is no other data in the Appalachian Mountains to confirm that this is common. An explanation for this may be the morphology of *Kalmia latifolia*. In some cases Ericaceae may have been missed because of their small size when they are young. At Brush Mountain and Reddish Knob there were no individuals that were younger than four years old. It is possible that younger vegetatively reproduced shrubs may not have breached the duff layer, and therefore were not collected. Maybe the high establishment for the 1990s was caused by mature branches being knocked over and buried by the duff and what would have been stems a decade ago were later considered individual shrubs. Or it could be possible that this last decade had less establishment than previous decades, for reasons not readily apparent.

It appears that the 1990s were a period of high shrub establishment at all the sites except for Reddish Knob. At Reddish Knob shrub thickets may have already reached an equilibrium, with such dense levels of Ericaceae that even they are not able to survive under the thick duff and low light conditions. Reddish Knob age structures showing more mid-aged shrubs than young shrubs provide evidence of this reduction in regeneration and recruitment. If this site continues without fire intense enough to remove Ericaceae and there are no other mechanisms to reduce the shrub layer, I suspect

that there will be little to no canopy regeneration. Although there are many studies about inhibition of canopy species under a shrub thicket, there are none that account for the age of the thickets and the history of fire. Whittaker (1956) suggested that sites with ericaceous shrubs that were highly stable may result in shrub heaths. Once overstory species die they may not be able to regenerate and Reddish Knob may turn into a heath bald, which has occurred in other ecosystems (Mallik 2003).

Clonal groups were difficult to characterize. *Kalmia latifolia* roots can be buried deep underground, and the morphology of ericaceous shrubs makes it hard to differentiate between an individual that came from a seed or an individual that established vegetatively. Because clonal shrubs are capable of naturally producing potentially independent offspring by means of vegetative growth (Van Groenendael et al. 1996) many of the individual clonal groups may have been connected deeper underground. The underground rhizomes of a congener, *Kalmia angustifolia*, can grow more than a meter in one growing season. They produce many vegetative sprouts (Mallik 1993) that can be hard to connect to the parent plant after years of being buried by the thick duff layer. There is no research about the vegetative growth of *Kalmia latifolia*, but field observations suggest that these methods of reproduction are similar to *Kalmia angustifolia* (Figure 3). In the field branch layering was common especially on Goldmine Trail where SPB outbreaks had caused tree fall resulting in *Kalmia latifolia* being pushed over. Once a branch is on the ground it is buried by leaf litter and will start to produce clonal shrubs from the fallen branches. When collecting data it became evident that the connectivity between individuals was expansive. It is likely that

estimates about vegetative reproduction are conservative. If vegetative reproduction could not be confirmed then shrubs are not reported as members of a clonal group. Across all sites there is very little evidence of vegetative reproduction before the last major fire. This could be a useful indicator to determine if fire affected a particular stand. This indicator could only be used if the fire occurred within the life span of the shrub.

McGee and Smith (1967) report that without disturbances ericaceous shrub thickets are not expanding. Their research suggests that management of these thickets could be postponed because of the stability of thickets. Some research reports that the ericaceous shrub layer sprouts heavily after fire, explaining that fire may worsen the density of ericaceous shrubs if not burned frequently (Arthur et al. 1998). These results show that without fire ericaceous shrubs are spreading vegetatively and management of the ericaceous shrub layer is indeed urgent.

Ducey et al. (2006) and Elliot et al. (1999) look at how recent fires influence ericaceous shrubs. Ducey et al. (2006) found that after a fire event the density of shrubs increased, and Elliot et al. (1999) saw a decrease in ericaceous shrubs. This research suggests that fire likely has a greater influence on basal area of ericaceous shrubs than density. This is most clearly shown at Griffith Knob. Although the number of stems may increase after a mild intensity fire the basal area and stature is reduced after a fire event so that canopy species are able to establish and possibly outcompete small statured Ericaceae.



The pre-burn vegetation of the study sites was likely influenced by the historic disturbance regime (Egler 1954). This research does not aim to discover the pre-fire suppression vegetation composition but to understand the present ericaceous shrub layer under a regime of fire suppression on sites with unique fire histories. This thesis provides an age structure of current ericaceous composition. There is a significant difference between shrub cover at these four sites. The environmental variables causing these differences are presumably numerous, however, the fire regime does seem to have had a noticeable influence on the shrub cover at these four sites. The low density and basal area of ericaceous shrubs at Griffith Knob coincides with my hypothesis that fire reduces shrubs in the Appalachian Mountains. The re-introduction of fire at Griffith Knob in the 1980s seems to have kept shrub cover low at this site. The last frequent fires occurred in the 1910s at Reddish Knob, this may have influenced the high shrub cover and older age structure of this site. Brush Mountain and Goldmine Trail have had comparable fire histories since fire suppression began but Blount County where Goldmine Trail is located has had many SPB outbreaks over the past 50 years (Forest Health Atlas 2010), if these outbreaks affected the pines at the Goldmine Trail site, they may have encouraged shrub establishment. After the cessation of fire on each site the shrub layer started to increase (Figure 28) and without anything to inhibit shrubs from establishing they have continued to occupy more topographic positions with higher coverage.

### Fire Effects

When closely examining cross sections to determine dates of establishment it seemed that many cross sections had fire scars (Figure 31). In some cases, when

comparing the dates of scars to fire events, scarring occurred at the same time fire events were recorded in the dendroecological record. It has previously been assumed or observed that when a fire burned the landscape that shrubs were removed from the surface, only later to repopulate by re-sprouting (Lafon and Kutac 2003). This data suggests otherwise. On Reddish Knob, transect two, at the mid-slope plot there were three cross sections out of 49 collected with scars that correspond to a 1985 isolated fire recorded on this site. This was the most recent fire on the site and only a single tree was scarred. Fire did not seem to have the effect of removing ericaceous shrubs at this site. This may have been the result of a mild intensity fire. If fires occur that are mild they may not burn hot enough to scar pines with thick bark. It is possible that they may even be mild enough to leave ericaceous shrubs intact. Apparently the fire intensity was low enough to scar some of the shrubs but not kill them. Because of the mild intensity of many prescribed burns they may not be effective in removing the ericaceous shrub layer in the future. Fire effects need to be closely monitored to assure goals of prescribed burning are reached.

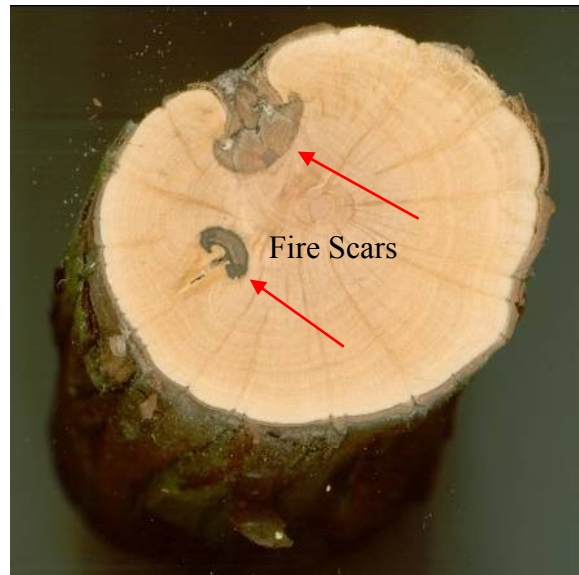


Figure 31. Fire scarred *Kalmia latifolia*. Cross section with two potential fire scars. These scars coincide with a mild isolated fire event.

### Biotic Influences

Although the relationship between establishment residuals and SPB was not significant, the data I used may not have been useful in showing any relationship with age structure residuals. The archive of SPB outbreaks extended back to the 1960s and ended in 2001. Only the four decades with a record of SPB could be used for correlation analysis. The county wide data had a low spatial resolution. This may be the reason why non-significant results were reached. While conducting fieldwork, observations were made that indicated that SPB had a positive influence on the ericaceous shrub layer at Goldmine Trail. In canopy gaps that had been caused by SPB there were often many *Kalmia latifolia* establishing, and few *Pinus spp.* observed re-establishing (Figure 32). Other research suggests that in a fire limited environment SPB can increase shrub

establishment and density (Lafon and Kutac 2003). SPB has no significant relationship to shrub increases that could be detected but observational evidence seems to suggest that it may influence new Ericaceae establishment. Recent data about SPB along with establishment dates with higher spatial resolution would be useful for understanding the influence of this disturbance.

My field observations also show that *Tsuga canadensis* had been affected by hemlock wooly adelgid at Goldmine Trail. Infestation only occurred at the lower slope plots where *Tsuga canadensis* was found. Although I did not perform any analysis with the influences of hemlock wooly adelgid and shrub establishment it is another disturbance to consider when analyzing shrub encroachment and is a question worth pursuing in future research

#### Role of Climate

Although it is likely that climate is a contributing variable to shrub establishment, there was no significant relationship detected in this research. Focus on climate change draws attention away from smaller localized issues that can be more easily controlled by land managers. Issues such as fire frequency and localized disturbance events seem to contribute more to shrub establishment and encroachment than larger scale influences such as PDSI and climate change.



Figure 32: *Kalmia latifolia* replacing yellow pine. A pine tree killed by southern pine beetle with *Kalmia latifolia* seedlings underneath.

## CHAPTER VI

### CONCLUSIONS

Ericaceous shrubs can have important influences on forest structure and function (Phillips and Murdy 1985; Clinton et al. 1994; Hedman and Van Lear 1995; Chastain and Townsend 2008; Beckage et al. 2000). Although ericaceous shrubs were a part of the landscape before fire suppression (Figure 33), the increase that has occurred under fire suppression has altered the dynamics of xeric pine-oak forests (Figure 34). Reddish Knob is an example of how fire suppression can increase the shrub layer, possibly causing irreversible modifications to the landscape.

Historically in these stands after a fire event adult pines were left standing and pine cones would be opened, allowing recruitment (Williams and Johnson 1992). Fire intolerant understory trees, shrubs and juveniles plants were killed (Zobel 1969). Many of the understory trees and shrubs that had a sprouting mechanism would spend energy vegetatively reproducing in stands with pine seedlings and adult pines between fires (Bond and Midgley 2001). In a high frequency fire regime, spending resources to reproduce underground parts that allowed for quick sprouting after fire may have been crucial to ericaceous shrubs maintaining their presence on the landscape.

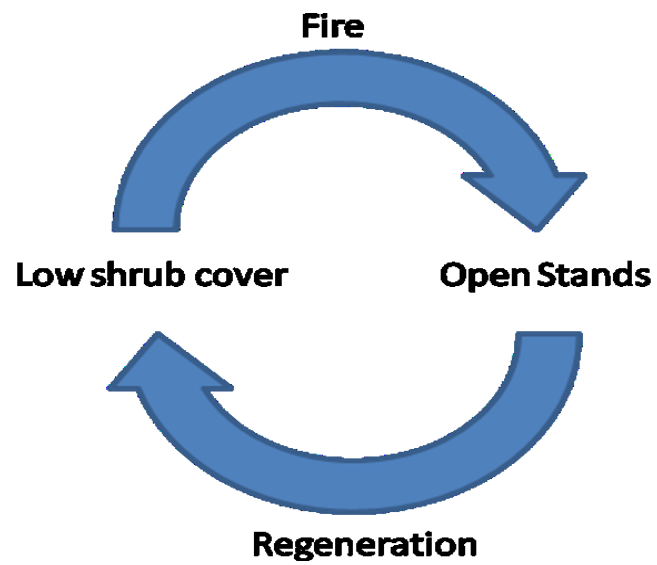


Figure 33: Conceptual model of the historical interaction between fire and shrub cover. Initially a stand would have relatively low shrub cover, then a fire would occur causing nearly open stands. Regeneration of the shrub layer would result in the site returning to an area with low shrub cover.

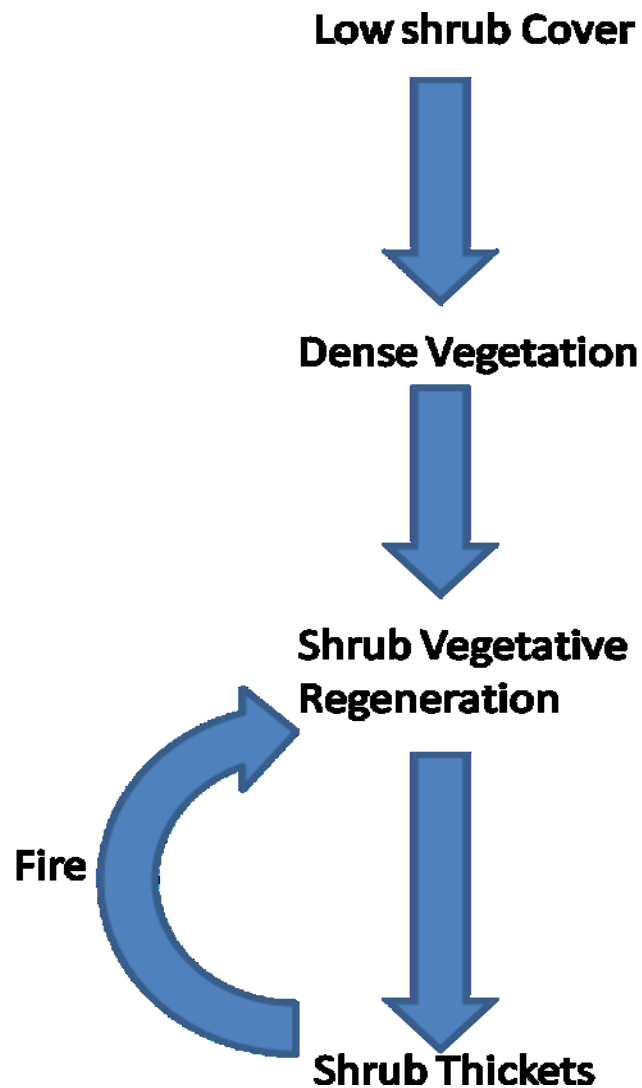


Figure 34: Conceptual model predicting future interactions between fire and shrub cover. A stand would begin with low shrub cover than as time since fire increased so would the density of vegetation. Shrubs would continue to increase, primarily through vegetative regeneration. Eventually shrub thickets would form inhibiting canopy regeneration. By this time fire would not be able to promote canopy establishment and shrub thickets would remain on the landscape.



The amount of understory trees and shrubs presumably were low with frequent fires. If Ericaceae were not removed they may be prevented from expending resources on regeneration or encroachment into new areas (Dobbs and Parker 2004). Without frequent fires adult pines continue to grow, but not reproduce, while fire intolerant species like ericaceous shrubs can reproduce vegetatively and through seed recruitment (Arthur 1998). Stands would become more dense and lower light conditions would inhibit new pines from establishing (Lafon et al. 2007). Fire intolerant hardwoods would start to encroach. In this environment shrubs could continue to reproduce in the low light environment, taking up more root space and altering duff layer characteristics (Monk et al. 1985). Pine seedlings will likely start to decrease because fires are no longer reducing the duff layer providing the bare soil required for establishment (Williams and Johnson 1992). Pines and other canopy species would eventually grow old and die. Ultimately shrubs would begin to dominate the landscape. Even after fires occurred and both the mature and juvenile ericaceous shrubs were removed from the surface they could re-sprout vegetatively (Brose and Waldrop 2010). With canopy species un-able to establish because of the extensive root system they could inhibit other species from invading, effectively becoming a shrub heathland.

Many non-fire disturbances that have occurred over the past century are beneficial to the shrub layer. Disturbances that reduce canopy cover and are beneficial to the shrub layer include but are not limited to ice storms, southern pine beetle attack, hemlock wooly adelgid invasion, gypsy moth invasion and wind storms. These

disturbances may remove canopy species or knock over ericaceous shrubs increasing encroachment.

When evaluating disturbances it is important to consider their effects on the ericaceous shrub layer, which are sometimes ignored. Cross-scale interactions between disturbances are likely (Peters et al. 2004). For example, when disturbances at one scale affect disturbances at larger or smaller scales it could result in increases or decreases of ericaceous shrubs. Without fire there are increases in SPB outbreaks (Knebel and Wentworth 2007). Fire suppression seems to be positively influencing the ericaceous shrub layer, while smaller scale SPB disturbances may be more severe as a result of fire suppression also causing increases in the shrub layer. The interaction between fire and SPB on different scales may cause shrub establishment to increase at a higher rate than in stands influenced by only one of these disturbances. The increase of the shrub layer could cause the duff layer to increase making it more difficult to burn. This would only cause higher shrub establishment. On the other hand an increase in fire that reduces the shrub layer, removes the duff layer and reduces susceptibility of pine to SPB will likely cause the ericaceous shrub layer to decrease establishment.

In a xeric pine-oak forest with a frequent fire regime the process of succession is usually re-initiated frequently enough to maintain the shade intolerant fire resistant pines. In a fire suppressed environment ericaceous shrubs can follow an inhibition pathway not allowing pines that were historically found on this site to establish once Ericaceae reach a certain density (Connell and Slayter 1977). The ericaceous shrubs populate the landscape after fire and inhibit other species from establishing. This

successional pathway usually occurs during secondary succession where a species, often shrubs, that reproduce vegetatively prevent other species from establishing. If shrub heathlands are truly self-replacing then stability or equilibrium is reached after a long period of fire suppression where shrubs outcompete other species.

At the most basic level this study is addressing the possibility of succession in a fire suppressed xeric-pine oak forest into areas dominated by ericaceous shrubs. The lack of fire on sites where *Kalmia latifolia* or other Ericaceae are abundant could result in a stable community composed primarily of ericaceous shrubs (Brose and Waldrop 2010). By looking at fire histories we can get an understanding about the past processes that contributed to the patterns of ericaceous shrub development. An understanding of past processes that have led to the current composition of vegetation can be important when trying to reach future goals for vegetation composition. Reduction of the ericaceous shrub layer is urgent if forests are going to be restored back to functioning pine forests. I suspect that it may be too late for fire alone to reduce shrub cover on the densest sites, while on others it is of the utmost urgency to begin burning with the appropriate intensity, frequency and seasonality.

## REFERENCES

- Abrams, M. D. 1992. Fire and the development of oak forests in eastern North-America, oak distribution reflects a variety of ecological paths and disturbance conditions. *Bioscience* 42: 346-353.
- Abrams, M. D. and G. J. Nowacki. 2008. Native Americans as active and passive promoters of mast and fruit trees in the eastern USA. *Holocene* 18:1123-1137.
- Aldrich, S. R., C. W. Lafon, H. D. Grissino-Mayer, G. G. DeWeese and J. A. Hoss. 2010. Three centuries of fire in montane pine-oak stands on a temperate forest landscape. *Applied Vegetation Science* 13:36-46.
- Aldrich, S. R. 2011. Fire Regimes and Successional Dynamics of Pine and Oak Forests in the Central Appalachian Mountains. Unpublished Dissertation. Texas A&M University.
- Anagnostakis, S. L. 1987. Chestnut Blight: The Classical Problem of an Introduced Pathogen. *Mycologia* 79:23-37.
- Anderson Jr, J. and F. Egler. 1988. Patch studies in the stability of non-diversity: *Dennstaedtia*, *Solidago*, *Spiraea*, *Kalmia*. *Phytologia* 64:349-364.
- Arthur, M. A., R. D. Paratley and B. A. Blankenship. 1998. Single and repeated fires affect survival and regeneration of woody and herbaceous species in an oak-pine forest. *Journal of the Torrey Botanical Society* 125:225-236.
- Bailey, R.G. 1980. Description of the Ecoregions of the United States. 1<sup>st</sup> ed. United States Department of Agriculture.
- Baker, T. T. and D. H. Van Lear. 1998. Relations between density of rhododendron thickets and diversity of riparian forests. *Forest Ecology and Management* 109:21-32.
- Barden, L.S. 2000. Population maintenance of *Pinus pungens* Lam.(Table Mountain pine) after a century without fire. *Natural Areas Journal* 20:227-233.
- 1976. Pine reproduction in thompson river watershed, north-carolina. *Journal of the Elisha Mitchell Scientific Society* 92:110-113.
- Beckage, B., J. S. Clark, B. D. Clinton and B. L. Haines 2000. A long-term study of tree seedling recruitment in southern Appalachian forests: the effects of canopy gaps and shrub understories. *Canadian Journal of Forest Research* 30:1617-1631.

- Beier, C. M., J. L. Horton, J. F. Walker, B. D. Clinton and E. T. Nilsen. 2005. Carbon limitation leads to suppression of first year oak seedlings beneath evergreen understory shrubs in Southern Appalachian hardwood forests. *Plant Ecology* 176:131-142.
- Bolstad, P. V. and W. T. Swank. 1997. Cumulative impacts of landuse on water quality in a southern Appalachian watershed. *Journal of the American Water Resources Association* 33:519-533.
- Bond, W. J. and J. E. Keeley. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* 20:387-394.
- Bond, W. J. and J. J. Midgley. 2001. Ecology of sprouting in woody plants: the persistence niche. *Trends in Ecology and Evolution* 16:45-51.
- Boring, L. R., C. D. Monk and W. T. Swank. 1981. Early regeneration of a clear-cut Southern Appalachian forest. *Ecology* 62:1244-1253.
- Bowman, I. 1911. *Forest physiography*. 1<sup>st</sup> ed. New York: John Wiley and Sons.
- Braun, E. 2001. *Deciduous Forests of Eastern North America*. Caldwell, NJ: Blackburn Press.
- Brose, P., T. Schuler, D. Van Lear and J. Berst. 2001. Bringing fire back - The changing regimes of the Appalachian mixed-oak forests. *Journal of Forestry* 99:30-35.
- Brose, P. H. and T. A. Waldrop. 2010. A dendrochronological analysis of a disturbance-succession model for oak-pine forests of the Appalachian Mountains, USA. *Canadian Journal of Forest Research*, 40:1373-1385.
- Brown, J. R. and S. Archer. 1999. Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology* 80:2385-2396.
- Chastain, R. A., W. S. Currie and P. A. Townsend. 2006. Carbon sequestration and nutrient cycling implications of the evergreen understory layer in Appalachian forests. *Forest Ecology and Management* 231:63-77.
- Chastain, R. A. and P. A. Townsend. 2008. Role of evergreen understory shrub layer in the forests of the central Appalachian Highlands. *Journal of the Torrey Botanical Society* 135:208-223.

- Clinton, B. D. 2003. Light, temperature, and soil moisture responses to elevation, evergreen understory, and small canopy gaps in the southern Appalachians. *Forest Ecology and Management* 186:243-255.
- Clinton, B. D. and C. R. Baker. 2000. Catastrophic windthrow in the southern Appalachians: characteristics of pits and mounds and initial vegetation responses. *Forest Ecology and Management* 126:51-60.
- Clinton, B. D. and L. R. Boring. 1994. Regeneration patterns in canopy gaps of mixed-oak forests of the southern Appalachians - influences of topographic position and evergreen understory. *American Midland Naturalist* 132:308-319.
- Clinton, B. D., J. M. Vose and W. T. Swank. 1993. Site preparation burning to improve southern appalachian pine hardwoods - vegetation composition and diversity of 13-year-old stands. *Canadian Journal of Forest Research* 23:2271-2277.
- Cook, Will. Trees, Shrubs, and Woody Vines of North Carolina. <http://www.duke.edu/~cwcook/trees/>. (Last Accessed 11/15/10).
- Connell, J. H. and R. O. Slatyer. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. *American Naturalist* 111:1119-1144.
- Daniels, L. D. and T. T. Veblen. 2004. Spatiotemporal influences of climate on altitudinal treeline in northern Patagonia. *Ecology* 85:1284-1296.
- Day, F. P. and C. D. Monk. 1974. Vegetation patterns on a Southern Appalachian watershed. *Ecology* 55:1064-1074.
- Delcourt, H. R. and P. A. Delcourt. 1997. Pre-Columbian Native American use of fire on southern Appalachian landscapes. *Conservation Biology* 11:1010-1014.
- 2000. *North American terrestrial vegetation*. Ed. Barbour, M. and W. Billings. Cambridge, MA: Cambridge University Press.
- Delcourt, P. and H. Delcourt. 2008. *Prehistoric Native Americans and Ecological Change: Human Ecosystems in Eastern North America since the Pleistocene*. Cambridge, MA: Cambridge University Press.
- DeWeese, G. 2007. Past fire regimes of Table Mountain pine (*Pinus pungens* L.) stands in the central Appalachian Mountains, Virginia, U.S.A. Unpublished Dissertation. University of Tennessee.

- Dietze, M. C. and J. S. Clark. 2008. Changing the gap dynamics paradigm: Vegetative regeneration control on forest response to disturbance. *Ecological Monographs* 78:331-347.
- Dobbs, M. M. and A. J. Parker. 2004. Evergreen understory dynamics in Coweeta Forest, North Carolina. *Physical Geography* 25:481-498.
- Doren, R. F., W. J. Platt and L. D. Whiteaker. 1993. Density and size structure of slash pine stands in the everglades region of south florida. *Forest Ecology and Management* 59:295-311.
- Ducey, M. J., W. K. Moser and P. M. S. Ashton. 1996. Effect of fire intensity on understory composition and diversity in a Kalmia-dominated oak forest, New England, USA. *Plant Ecology* 123:81-90.
- Elliott, K. J., R. L. Hendrick, A. E. Major, J. M. Vose and W. T. Swank. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. *Forest Ecology and Management* 114:199-213.
- Elliott, K. J., S. L. Hitchcock and L. Krueger. 2002. Vegetation response to large scale disturbance in a southern Appalachian forest: Hurricane Opal and salvage logging. *Journal of the Torrey Botanical Society* 129:48-59.
- Eppard, H. R., J. L. Horton, E. T. Nilsen, P. Galusky and B. D. Clinton. 2005. Investigating the allelopathic potential of Kalmia latifolia L. (Ericaceae). *Southeastern Naturalist* 4:383-392.
- Fenneman, N. 1938. *Physiography of eastern United States*. New York, London: McGraw-Hill Book Company, inc.
- Forbush, E., C. Fernald and M. S. B. o. Agriculture. 1896. *The gypsy moth: Porthetria dispar (Linn.). A report of the work of destroying the insect in the commonwealth of Massachusetts, together with an account of its history and habits both in Massachusetts and Europe*. Boston, MA: Wright and Potter Printing Co.
- Grissino-Mayer, H. D., W. H. Romme, M. L. Floyd and D. D. Hanna. 2004. Climatic and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. *Ecology* 85:1708-1724.
- Harmon, M. E. 1982. Decomposition of standing dead trees in the southern Appalachian Mountains. *Oecologia* 52:214-215.

- 1984. Survival of trees after low-intensity surface fires in Great Smoky Mountains National-Park. *Ecology* 65:796-802.
- Harrod, J., P. White and M. Harmon. 1998. Changes in xeric forests in western Great Smoky Mountains National Park, 1936-1995. *Castanea* 63:346-360.
- Harrod, J. C., M. E. Harmon and P. S. White. 2000. Post-fire succession and 20th century reduction in fire frequency on xeric southern Appalachian sites. *Journal of Vegetation Science* 11:465-472.
- Hedman, C. W. and D. H. Vanlear. 1995. Vegetative structure and composition of southern appalachian riparian forests. *Bulletin of the Torrey Botanical Club* 122:134-144.
- HilleRisLambers, J. and J. S. Clark. 2003. Effects of dispersal, shrubs, and density-dependent mortality on seed and seedling distributions in temperate forests. *Canadian Journal of Forest Research* 33:783-795.
- Hinman, S., J. S. Brewer and S. W. Ashley. 2008. Shrub seedling establishment is limited by dispersal, slow growth, and fire in two wet pine savannahs in Mississippi. *Natural Areas Journal* 28:37-43.
- Holzmueeller, E., S. Jose and M. Jenkins. 2009. The Response of Understory Species Composition, Diversity, and Seedling Regeneration to Repeated Burning in Southern Appalachian Oak-Hickory Forests. *Natural Areas Journal* 29:255-262.
- Hooper, R. 1969. *Prescribed Burning For Laurel and Rhododendron Control in the Southern Appalachians*. US Forest Service Research Note SE-116.
- Hoss, J. A., C. W. Lafon, H. D. Grissino-Mayer, S. R. Aldrich and G. G. DeWeese. 2008. Fire history of a temperate forest with an endemic fire-dependent herb. *Physical Geography* 29:424-441.
- Johnson, C. G. 1990. Response of vegetation to livestock impacts on green fescue sites in the whitebark-pine ecosystem. In Proceedings - Symposium on Whitebark Pine Ecosystems: Ecology and Management of a High-Mountain Resource, eds. W. C. Schmidt and K. J. McDonald, 294-296. Ogden: US Department of Agriculture, Forest Service Intermountain Research Station.
- Keever, C. 1953. Present composition of some stands of the former oak-chestnut forest in the southern blue ridge mountains. *Ecology* 34:44-54.



Kentucky Geological Survey, Publications, Volume 2 Part 3, 1915.

Knebel, L. and T. R. Wentworth. 2007. Influence of fire and southern pine beetle on pine-dominated forests in the Linville Gorge Wilderness, North Carolina. *Castanea* 72:214-225.

Lafon, C. W., J. A. Hoss and H. D. Grissino-Mayer. 2005. The contemporary fire regime of the central Appalachian Mountains and its relation to climate. *Physical Geography* 26:126-146.

Lafon, C. W. and M. Kutac. 2003. Effects of ice storms, southern pine beetle infestation, and fire on table mountain pine forests of southwestern Virginia. *Physical Geography* 24: 502-519.

Lafon, C. W., J. D. Waldron, D. M. Cairns, M. D. Tchakerian, R. N. Coulson and K. D. Klepzig. 2007. Modeling the effects of fire on the long-term dynamics and restoration of yellow pine and oak forests in the southern Appalachian Mountains. *Restoration Ecology* 15:400-411.

Lambert, R. 1961. Logging the Great Smokies, 1880-1930. *Tennessee Historical Quarterly* 20:350-363.

Lei, T. T., S. W. Semones, J. F. Walker, B. D. Clinton and E. T. Nilsen. 2002. Effects of *Rhododendron maximum* thickets on tree seed dispersal, seedling morphology, and survivorship. *International Journal of Plant Sciences* 163:991-1000.

Luteyn, J. L., W. S. Judd, S. P. V. Kloet, L. J. Dorr, G. D. Wallace, K. A. Kron, P. F. Stevens and S. E. Clemants. 1996. Ericaceae of the Southeastern United States. *Castanea* 61:101-144.

Mallik, A. U. 1987. Allelopathic Potential of *Kalmia-Angustifolia* to Black Spruce (*Picea-Mariana*). *Forest Ecology and Management* 20:43-51.

--- .1993. Ecology of a forest weed of newfoundland - vegetative regeneration strategy of *kalmia-angustifolia*. *Canadian Journal of Botany* 71:161-166.

--- 2003. Conifer regeneration problems in boreal and temperate forests with ericaceous understory: Role of disturbance, seedbed limitation, and keystone species change. Critical Reviews in *Plant Sciences* 22:341-366.

McClure, M. and C. A. E. Station. 1987. *Biology and control of hemlock wooly adelgid*. Connecticut Agricultural Experiment Station.

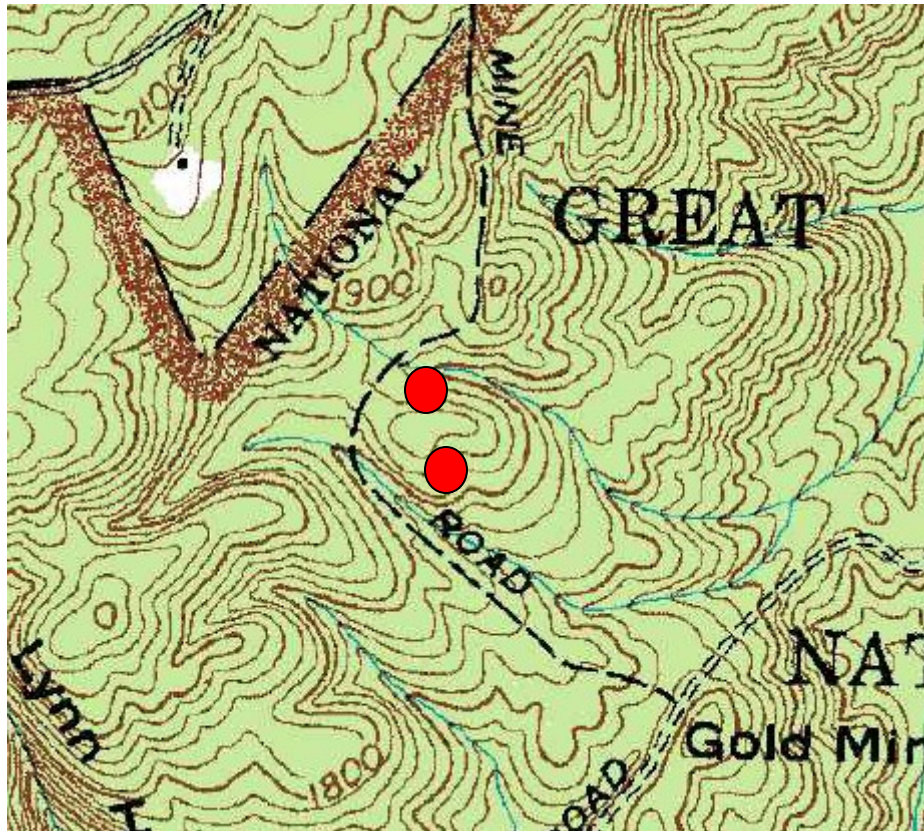
- McGee, C. E. and R. C. Smith. 1967. Undisturbed rhododendron thickets are not spreading. *Journal of Forestry* 65:334-335.
- Mickler, R. A. and S. Fox. 1998. *The productivity and sustainability of southern forest ecosystems in a changing environment*. New York: Springer.
- Monk, C. D., D. T. McGinty and F. P. Day, Jr. 1985. The Ecological Importance of *Kalmia latifolia* and *Rhododendron maximum* in the Deciduous Forest of the Southern Appalachians. *Bulletin of the Torrey Botanical Club* 112:187-193.
- National Climatic Data Center, Monthly Station Normals of Temperature, Precipitation, and Heating and Cooling Degree Days 1971 – 2000. Climatography of the United States no. 81.
- Nilsen, E. T., J. F. Walker, O. K. Miller, S. W. Semones, T. T. Lei and B. D. Clinton. 1999. Inhibition of seedling survival under *Rhododendron maximum* (Ericaceae): Could allelopathy be a cause? *American Journal of Botany* 86:1597-1605.
- Noble, I. R. and R. O. Slatyer. 1980. The Use of Vital Attributes to Predict Successional Changes in Plant Communities Subject to Recurrent Disturbances. *Vegetation* 43:5-21.
- Orwig, D. A. and D. R. Foster. 1998. Forest Response to the Introduced Hemlock Woolly Adelgid in Southern New England, USA. *Journal of the Torrey Botanical Society* 125:60-73.
- Ott, L. and M. Longnecker. 2010. *An Introduction to Statistical Methods and Data Analysis*. Belmont, CA: Brooks/Cole Cengage Learning.
- Parker, A. J. 1986. Persistence Of Lodgepole Pine Forests In The Central Sierra-Nevada. *Ecology* 67:1560-1567.
- Peters, D., R. Pielke, B. Bestelmeyer, C. Allen, S. Munson-McGee and K. Havstad. 2004. Cross-scale interactions, nonlinearities, and forecasting catastrophic events. *Proceedings of the National Academy of Sciences of the United States of America*, 101:15130-15135.
- Peters, D. P. C., B. T. Bestelmeyer, J. E. Herrick, E. L. Fredrickson, H. C. Monger and K. M. Havstad. 2006. Disentangling complex landscapes: New insights into arid and semiarid system dynamics. *Bioscience* 56:491-501.

- Peterson, G. D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. *Ecosystems* 5:329-338.
- Phillips, D. L. and W. H. Murdy. 1985. Effects of rhododendron (rhododendron-maximum l) on regeneration of southern appalachian hardwoods. *Forest Science* 31:226-233.
- Pyle, C. 1988. The Type and Extent of Anthropogenic Vegetation Disturbance in the Great Smoky Mountains before National-Park-Service Acquisition. *Castanea*, 53:183-196.
- Pyne, S. 1997. *Fire in America: A Cultural History of Wildland and Rural Fire*. Seattle, WA: University of Washington Press.
- Rice, E.L. 1984. *Allelopathy*. New York, NY: Academic Press.
- Rodgers, J. 1970. *The Tectonics of the Appalachians*. Hoboken, NJ: Wiley-Interscience.
- Schowalter, T. D., R. N. Coulson and D. A. Crossley. 1981. Role of southern pine-beetle dendroctonus-frontalis zimmerman (coleoptera, scolytidae) and fire in maintenance of structure and function of the southeastern coniferous forest. *Environmental Entomology*, 10:821-825.
- Shumway, D. L., M. D. Abrams and C. M. Ruffner. 2001. A 400-year history of fire and oak recruitment in an old-growth oak forest in western Maryland, USA. *Canadian Journal of Forest Research* 31:1437-1443.
- Shure, D. J., D. L. Phillips and P. E. Bostick. 2006. Gap size and succession in cutover southern Appalachian forests: an 18 year study of vegetation dynamics. *Plant Ecology* 185:299-318.
- Smith, G. F. and N. S. Nicholas. 1998. Patterns of overstory composition in the fir and fir-spruce forests of the Great Smoky Mountains after balsam woolly adelgid infestation. *American Midland Naturalist* 139:340-352.
- Van Auken, O. W. 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31:197-215.
- Van Groenendael, J.M., L. Klimes, J Kimesova and R.J.J. Hendriks. 1996. Comparative Ecology of Clonal Plants. *Philosophical Transactions: Biological Sciences* 351:1331-1339.

- Van Lear, D., T. Waldrop and S. F. E. Station. 1989. History, uses, and effects of fire in the Appalachians. US Dept. of Agriculture, Forest Service, Southeastern Forest Experiment Station.
- Vandermaast, D. B. and D. H. Van Lear. 2002. Riparian vegetation in the southern Appalachian mountains (USA) following chestnut blight. *Forest Ecology and Management* 155: 97-106.
- Waldrop, T. A., L. Brudnak and S. Rideout-Hanzak. 2007. Fuels on disturbed and undisturbed sites in the southern Appalachian Mountains, USA. *Canadian Journal of Forest Research* 37:1134-1141.
- Waldrop, T. A., N. T. Welch, P. H. Brose, K. J. Elliott, H. H. Mohr, E. A. Gray, F. H. Tainter, L. E. Ellis and U. U. U. USDA. 2000. Current research on restoring ridgetop pine communities with stand replacement fire. In Proceedings: Workshop on Fire, People, and the Central Hardwood Landscape, 103-109. Radnor: US Dept of Agriculture, Forest Service NE Experimental Station.
- Walker, J. F., O. K. Miller, T. Lei, S. Semones, E. Nilsen and B. D. Clinton. 1999. Suppression of ectomycorrhizae on canopy tree seedlings in *Rhododendron maximum* L-(Ericaceae) thickets in the southern Appalachians. *Mycorrhiza* 9:49-56.
- Whiteman, C. 2000. *Mountain meteorology: fundamentals and applications*. New York, NY. Oxford University Press.
- Whitney, H. E. and W. C. Johnson. 1984. Ice storms and forest succession in southwestern Virginia. *Bulletin of the Torrey Botanical Club*, 111:429-437.
- Whittaker, R. H. 1956. Vegetation of the Great Smoky Mountains. *Ecological Monographs* 26:1-69.
- 1962. Net production relations of shrubs in Great Smoky Mountains. *Ecology* 43:357-377.
- Williams, C. E. and W. C. Johnson. 1992. Factors affecting recruitment of *Pinus pungens* in the southern Appalachian Mountains. *Canadian Journal of Forest Research* 22:878-887.
- Zobel, D. B. 1969. Factors affecting distribution of *Pinus pungens*, an Appalachian endemic. *Ecological Monographs* 39:303-333.

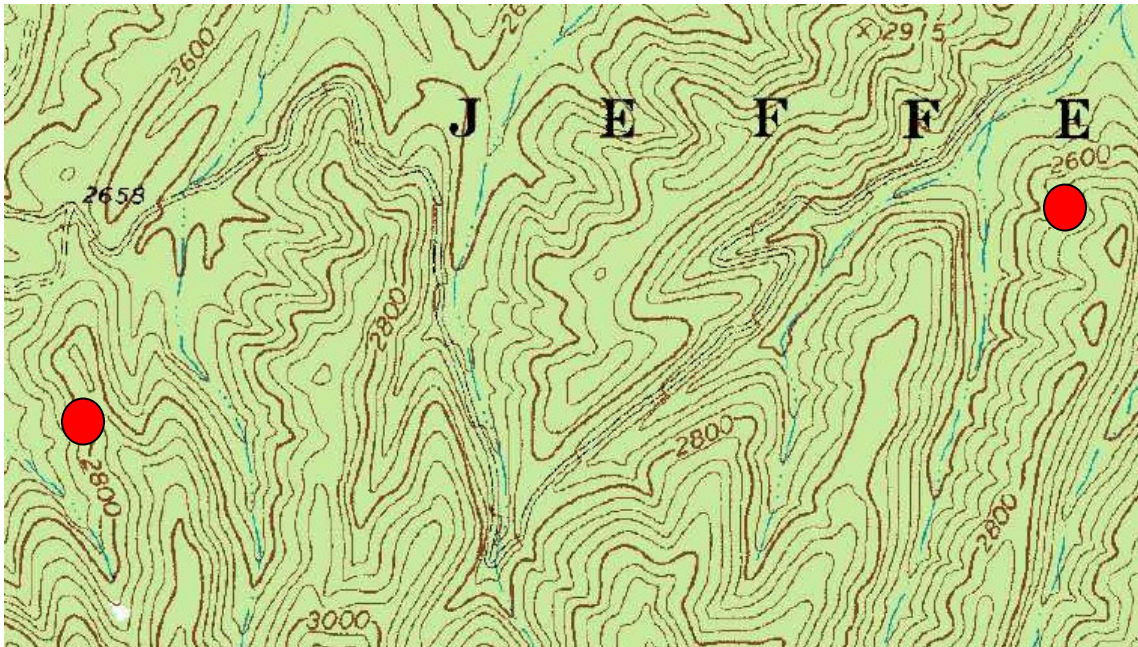
## APPENDIX

## Maps of study sites

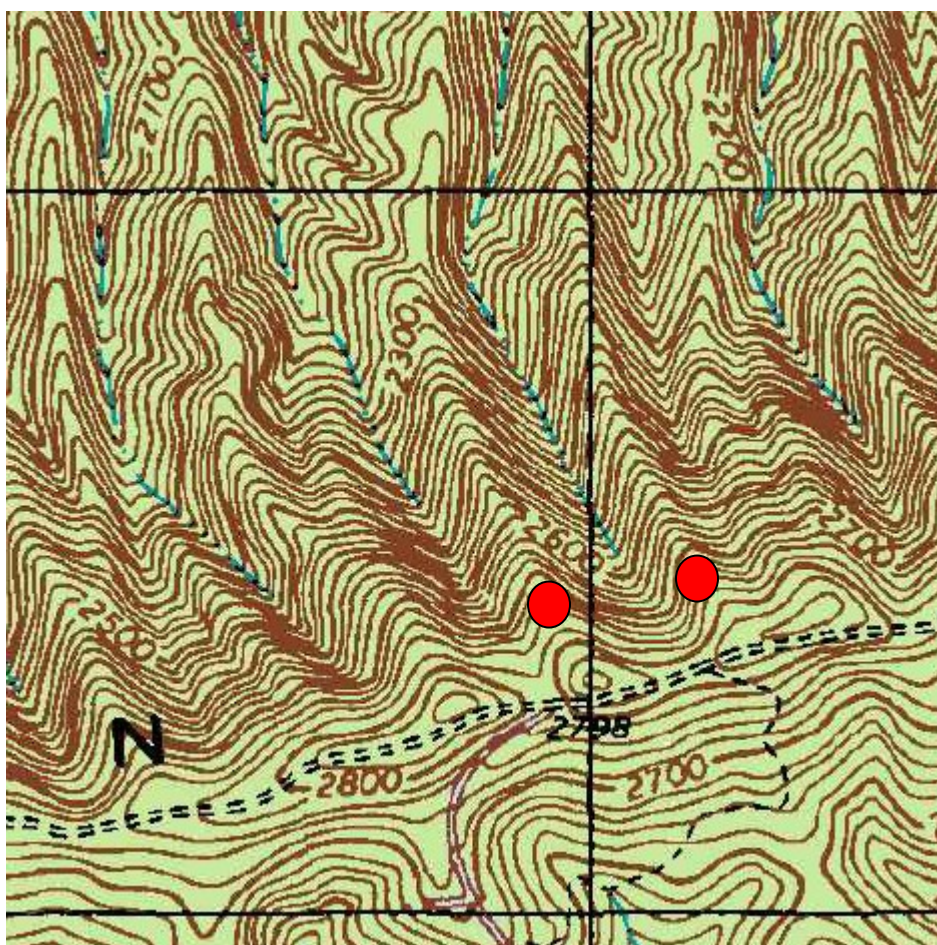


Map 1: Location of Goldmine Trail. Dots represent the location of transects.  
Blockhouse, TN topographic 7.5x7.5 quadrangle, 1:24,000.



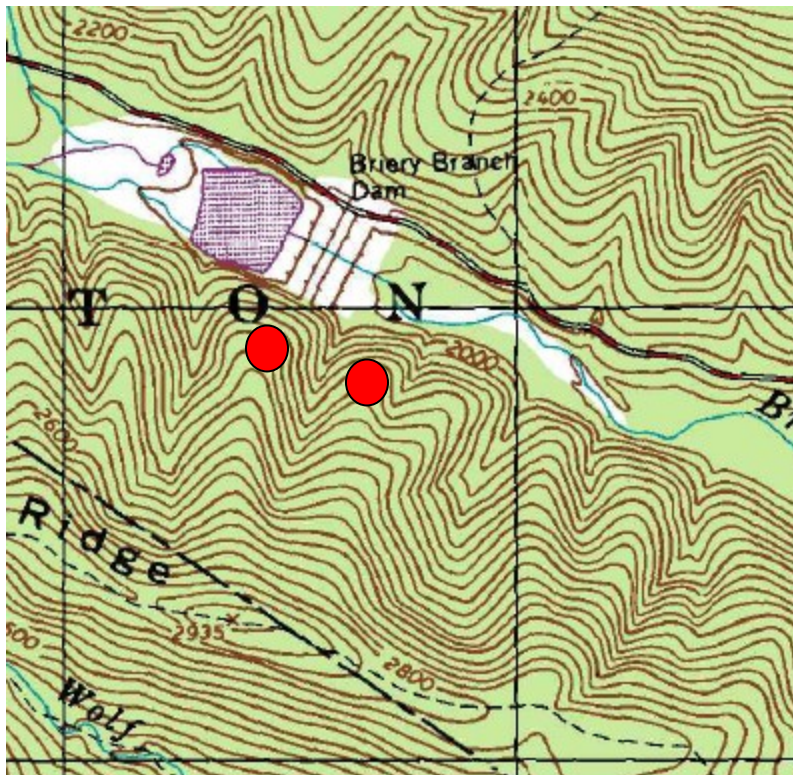


Map 2: Location of Griffith Knob. Dots represent the location of transects. Big Bend, VA 7.5x7.5 topographic quadrangle, 1:24,000.



Map 3: Location of Brush Mountain. Dots represent the location of transects. Newport, VA 7.5x7.5 topographic quadrangle, 1:24,000.





Map 4: Location of Reddish Knob. Dots represent the location of transects. Reddish Knob, VA 7.5x7.5 topographic quadrangle, 1:24,000.



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